

**INTERIM REPORT ON
VIABILITY CRITERIA FOR
WILLAMETTE AND LOWER COLUMBIA BASIN
PACIFIC SALMONIDS**

WILLAMETTE/LOWER COLUMBIA TECHNICAL RECOVERY TEAM

MARCH 31, 2003

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TABLE OF CONTENTS

List of Figures.....	vii
List of Tables	viii
Abbreviations and Acronyms	ix
1. Introduction.....	1
2. ESU-Level Viability	6
<i>Overview.....</i>	<i>6</i>
<i>Viability Criteria Approach.....</i>	<i>9</i>
<i>Number of Populations per Stratum.....</i>	<i>11</i>
<i>Selection of Populations in a Stratum</i>	<i>15</i>
Metapopulation Processes	15
Evolutionary Processes.....	16
Catastrophic Risk.....	16
Approach to Selecting Populations.....	17
<i>ESU-Level Viability Criteria and Strata Persistence.....</i>	<i>17</i>
<i>Recovery Strategy Criteria.....</i>	<i>18</i>
<i>Examples of Viable ESU Scenarios.....</i>	<i>19</i>
3. Integrating Attributes and Assessing Population Risk of Extinction.....	24
<i>Overview.....</i>	<i>24</i>
<i>Approach to Integrating Population Attributes.....</i>	<i>25</i>
4. Population Productivity and Abundance Criteria.....	27
<i>Overview.....</i>	<i>27</i>
Key Issues.....	27
Approaches Considered	28
<i>PVA Modeling</i>	<i>30</i>
Overview	30
Fitting Spawner Recruit Curves.....	33
<i>Two-Life-Stage Projection Model</i>	<i>34</i>
<i>Population Change Criteria.....</i>	<i>34</i>
Overview	34
Complications in Estimating Productivity Addressed with PCC	35
Example Output of Population Change Criteria.....	36
Conclusions Regarding PVA Modeling Approaches	37
<i>Historical Abundance.....</i>	<i>38</i>

Historical Abundance Estimates Using Broad-Scale Habitat Analysis	38
Historical Abundance Estimates Using HPVA	39
Risk Characterization.....	40
5. Juvenile Outmigrant Growth-Rate Criteria.....	41
<i>Definitions</i>	41
<i>Overview</i>	41
<i>Approaches Considered</i>	43
<i>Strategies Selected</i>	44
<i>Critical Uncertainties</i>	45
<i>Monitoring and Evaluation</i>	45
Risk Characterization.....	49
6. Within-Population Diversity Criteria	51
<i>Overview</i>	51
<i>Approaches Considered</i>	53
<i>Approach Selected</i>	54
<i>Within-Population Diversity Recovery Strategies</i>	55
<i>Critical Uncertainties</i>	56
<i>Monitoring and Evaluation</i>	57
<i>Risk Characterization</i>	58
7. Habitat Criteria.....	60
<i>Overview</i>	60
<i>Strategies Selected</i>	61
<i>Relationship of Habitat to Other Criteria</i>	62
<i>Critical Uncertainties</i>	63
<i>Future Development of Specific Habitat Criteria</i>	63
<i>Habitat Risk Characterization</i>	64
8. Within-Population Spatial Structure Criteria.....	67
<i>Overview</i>	67
<i>Approaches Considered</i>	69
<i>Strategies Selected</i>	72
<i>Risk Characterization</i>	73
<i>Critical Uncertainties</i>	74
<i>Monitoring and Evaluation</i>	76
9. Literature Cited	78

Appendix A	ESU Strata—Ecological Zone and Life-History Type
Appendix B	Summary of Core and Genetic Legacy Populations
Appendix C	Method for Integrating Attributes and Assessing Population Risk of Extinction
Appendix D	Population Change Criteria
Appendix E	Estimates of Environmental Variance for PCC Analysis
Appendix F	Concerns Raised about Population Change Criteria Approach
Appendix G	Exploration of Spawner-Recruit Analysis and Viability Criteria of WLC Populations
Appendix H	A Two-Stage Spawner-Recruit Model of Population Viability
Appendix I	Broad-Scale Habitat Analyses to Estimate Fish Densities for Viability Criteria
Appendix J	HPVA Results for Salmon and Steelhead Production in the Washington Lower Columbia Basins
Appendix K	Catastrophic Risk Assessment of Lower Columbia and Willamette River ESUs for Threatened Pacific Salmon
Appendix L	Maximum Temperature: Upper Optimal Temperature Limits for Salmonids in the Willamette and Lower Columbia Rivers

LIST OF FIGURES

Figure 1.1	Approach to ESU criteria	4
Figure 2.1	Probability of losing all the populations in a stratum within 100 years as a function of the initial number of populations, assuming populations are independent	12
Figure 2.2	EXAMPLE Lower Columbia chinook salmon ESU scenario	20
Figure 2.3	EXAMPLE Upper Willamette steelhead viable ESU scenario	21
Figure 4.1	Conceptual graph of the relationship between productivity, population size, and extinction risk	31
Figure 4.2	Different types of spawner-recruit curves	32
Figure 5.1	Hypothetical changes in spawner and smolt abundance in a 20-year cycle (sine-wave) of ocean survival.....	42
Figure 5.2.	Hypothetical changes in the maximum intra-annual variability allowed for detection of λ_{JOM} , given an increasing number of samples.....	47
Figure 5.3	Three JOM abundance estimate scenarios, showing the importance of intra-annual variability for trend estimation	48
Figure 8.1	Venn diagram of relationship between habitat and spatial structure	68

LIST OF TABLES

Table 2.1	Estimated number of historical demographically independent populations.....	7
Table 2.2	Estimated number of populations in different strata in the WLC domain.....	10
Table 2.3	Description of population persistence categories	13
Table 2.4	Stratum persistence categories based on averages of individual population risks	14
Table 2.5	Probability of achieving recovery of at least three populations if the probability of a successful population recovery attempt is 80%.....	18
Table 2.6	Random collection of population persistence categories used for example viable ESU scenarios	20
Table 2.7	EXAMPLE selection of populations needed above persistence category 3 for example viable ESU scenarios.....	22
Table 3.1	Description of population persistence categories	24
Table 3.2	Example “summary population profile table”	25
Table 4.1	Definitions of time intervals used in discussion of population productivity and abundance criteria.....	32
Table 4.2	Growth and abundance viability criteria expressed as growth rate	36
Table 4.3	PCC productivity criteria for conditions where hatchery fish are present or marine survival over the observation period differs from the long-term average	37
Table 5.1	Relationship between λ_{JOM} and population persistence category	49
Table 6.1	Risk characterization for within-population diversity	59
Table 7.1	Examples of habitat attributes and life-history stages for which each attribute is particularly critical	62
Table 7.2	Risk characterization for habitat criteria	66

ABBREVIATIONS AND ACRONYMS

BRT	biological review team
CWT	coded-wire tag
DIP	demographically independent population
EDT	ecosystem diagnosis and treatment
ESA	Endangered Species Act
ESU	evolutionarily significant unit
HPVA	habitat population viability analysis
λ_{JOM}	JOM growth rate
JOM	juvenile outmigrant
NWIFC	Northwest Indian Fisheries Commission
NMFS	National Marine Fisheries Service (also referred to as NOAA Fisheries)
PCC	population change criteria
PFC	properly functioning conditions
PVA	population viability analysis
QET	quasi-extinction threshold
TRT	technical recovery team
VSP	viable salmonid population
WDFW	Washington Department of Fish and Wildlife
WLC	Willamette/Lower Columbia

1. INTRODUCTION

In 1995, the National Marine Fisheries Service (NOAA Fisheries) announced that it would initiate a series of coastwide status reviews of anadromous salmonids in the states of Washington, Idaho, Oregon, and California. These status reviews resulted in the listing of several evolutionarily significant units (ESUs) of salmonids as threatened or endangered under the U.S. Endangered Species Act (ESA). Listed ESUs were organized into geographically proximate units, called recovery domains, as part of a multispecies approach that could address common regional recovery issues. This report focuses on the Willamette/Lower Columbia (WLC) domain.

The WLC domain contains five listed ESUs and one candidate ESU:

- Columbia River chum salmon (listed as threatened, 1999),
- Lower Columbia River steelhead (listed as threatened 1998),¹
- Lower Columbia River chinook salmon (listed as threatened 1999),
- Upper Willamette River steelhead (listed as threatened 1999),¹
- Upper Willamette River chinook salmon (listed as threatened 1999),
- Lower Columbia River coho salmon (candidate species 1997).

To obtain advice on technical issues related to recovery planning, NOAA Fisheries convened technical recovery teams (TRTs) in each recovery domain. The TRTs are composed of scientists from NOAA Fisheries; other federal, tribal, state and local agencies; academic institutions; and private consulting firms. A complete description of the TRT composition, tasks, relationship to ESA recovery planning, and operating principles can be found in the NMFS document *Recovery Planning Guidance for Technical Recovery Teams (TRTs)* (<http://research.nwfsc.noaa.gov/cbd/trt/about.htm>). The Willamette/Lower Columbia Technical Recovery Team (WLC-TRT) was established in May 2000. One of its first tasks was to provide technical information to support the development of delisting criteria. This report is a response to that task. The main text of this report is a consensus product of the TRT. The report contains a number of appendices in support of the main text, which are not TRT consensus products. The appendix authors are identified at the beginning of each appendix. Most of the appendices include individual TRT members as authors; the appendices were produced in coordination with and in support of the TRT.

Under the ESA, NOAA Fisheries must identify measurable and objective delisting criteria as part of recovery planning. The delisting criteria must describe the conditions under which a listed species or ESU is no longer in danger of extinction (endangered) or likely to become so in the foreseeable future (threatened). We define a viable ESU as one that is unlikely to be at risk of extinction. Ultimately, the crafting of delisting criteria requires the consideration of technical analyses relating to viability, which are contained in this document, and policy decisions such as acceptable levels of risk, which are not. This document presents the WLC-TRT's viability criteria guidelines. As with any scientific conclusions, it is anticipated that the recommendations in this document may be revised in the future based on new data or analysis. In

¹ Both anadromous forms of *Oncorhynchus mykiss* (steelhead trout) and resident forms *O. mykiss* (rainbow trout) often occur in the same river systems. The genetic and demographic relationships among these two life history types are poorly understood. In this document, we concentrate on criteria related to anadromous *O. mykiss*.

addition, the criteria need to describe viability conditions in a way that is usable by managers, thus revised viability criteria may be developed in the future in response to interaction with managers developing recovery goals.

The listed unit under the ESA for Pacific salmon is the ESU, and this is the unit that must be considered for delisting. Thus, delisting criteria must ultimately address the overall extinction risk of the ESU. In approaching the development of criteria for delisting ESUs we have relied on the language in the ESA, information described in the listing decision, concepts outlined in a report on viable salmonid populations (VSPs) by McElhany et al. (2000), which can be found online at <http://www.nwfsc.noaa.gov/pubs/tm/tm42/tm42.pdf>; and in published research describing salmon populations and their past or potential responses to changes in climate and ocean conditions (e.g., Bradford and Irving 2000).

The ESA lists five potential factors for decline that must be considered in species listing decisions (ESA Section 4.2.1):

1. the present or threatened destruction, modification, or curtailment of its habitat or range;
2. overutilization for commercial, recreational, scientific, or educational purposes;
3. disease or predation;
4. the inadequacy of existing regulatory mechanisms;
5. other natural or manmade factors affecting its continued existence.

The NMFS identified all five factors as contributing to the endangerment of Pacific salmonids. In considering how viability criteria might inform population delisting requirements, the TRT also considered all five factors for decline.

The TRT approach evaluates the extinction risks facing an ESU by assessing the viability of the individual populations within that ESU. The TRT identified population-level viability criteria based on:

1. a combination of the four population parameters identified in the VSP document—productivity, abundance, spatial structure, and diversity;
2. information about the habitat requirements of the listed salmon; and
3. the need to separate population responses to freshwater and estuarine habitat conditions from population responses to fluctuating marine and climatic conditions.

Building from these population-level criteria, the WLC-TRT developed the ESU criteria framework in Figure 1.1.

The approach depends first on identifying historical, demographically independent populations within each ESU. Population identification focuses on demographically independent units because many of the processes affecting extinction risk operate on this scale. (This concept is discussed in more detail in McElhany et al. 2002) A draft WLC-TRT document (Myers et al. 2002) estimates historical population boundaries for all five listed ESUs in the WLC domain. The populations identified in Myers et al. are used in this report.

The overall approach we have taken to establishing ESU viability criteria is outlined in Figure 1.1. In Section 2, we develop the ESU-level viability criteria by first dividing the ESU into groups of populations called strata. The strata are based on major life-history characteristics (e.g., spring versus fall chinook) and ecological zones. The ecological zones are relatively large

scale, with the Lower Columbia being divided into three ecological zones and the Willamette consisting of a single ecological zone. The strata represent major diversity components of the ESUs, and populations in the different strata are likely to be subject to different catastrophic events. Therefore, it is important for ESU viability to ensure a reasonably high probability of persistence for each stratum, so the ESU-level viability criteria are crafted as a function of the persistence probability of each stratum. The persistence probability of each stratum is in turn a function of the viability of its component populations, and in Section 2 we develop an algorithm for estimating how many and which populations need to have a given persistence probability in each stratum.

This approach requires some way to assess the overall persistence probability of individual populations. In assessing individual populations, the WLC-TRT developed guidelines for criteria about five attributes:

1. adult productivity and abundance,
2. juvenile outmigrant (JOM) growth rate,
3. within-population diversity,
4. habitat, and
5. within-population spatial structure.

In order for a population to be considered viable, it would have to meet the criteria for all five attributes. There is necessarily some redundancy built into these criteria guidelines. In a simple world, one criterion would suffice, but there are many potential scenarios in which any four of the above criteria might lead to a false conclusion that a population's extinction risk was low enough for it to be considered viable. For example, in a scenario in which adult salmon escapements are increased under favorable ocean conditions some criteria may be met (e.g., adult productivity and abundance, within-population spatial structure, within-population diversity, and habitat), yet JOM criteria would indicate that actual freshwater production was not yet high enough to delist. Alternatively, habitat criteria might not be met when all four fish performance criteria are met, indicating that populations were in a period of favorable ocean and climatic conditions, but that freshwater and estuarine habitats were not yet of sufficient quality to support the population during less favorable ocean and climate conditions.

In Section 3, we propose several approaches for integrating each individual attribute into an overall assessment of population persistence probability. In Sections 4 through 8, we describe criteria associated with each individual population attribute.

This report does not describe what actions need to be taken to restore salmonid ESUs. It is limited to describing measurable and objective attributes of VSPs and ESUs. For example, some reasons that ESA-listed salmonid populations may be threatened with extinction include the following:

- Density-independent survival is too low for populations to replace themselves.
- The carrying capacity of the population is low enough for density-dependent processes to keep the population at critically low abundance.

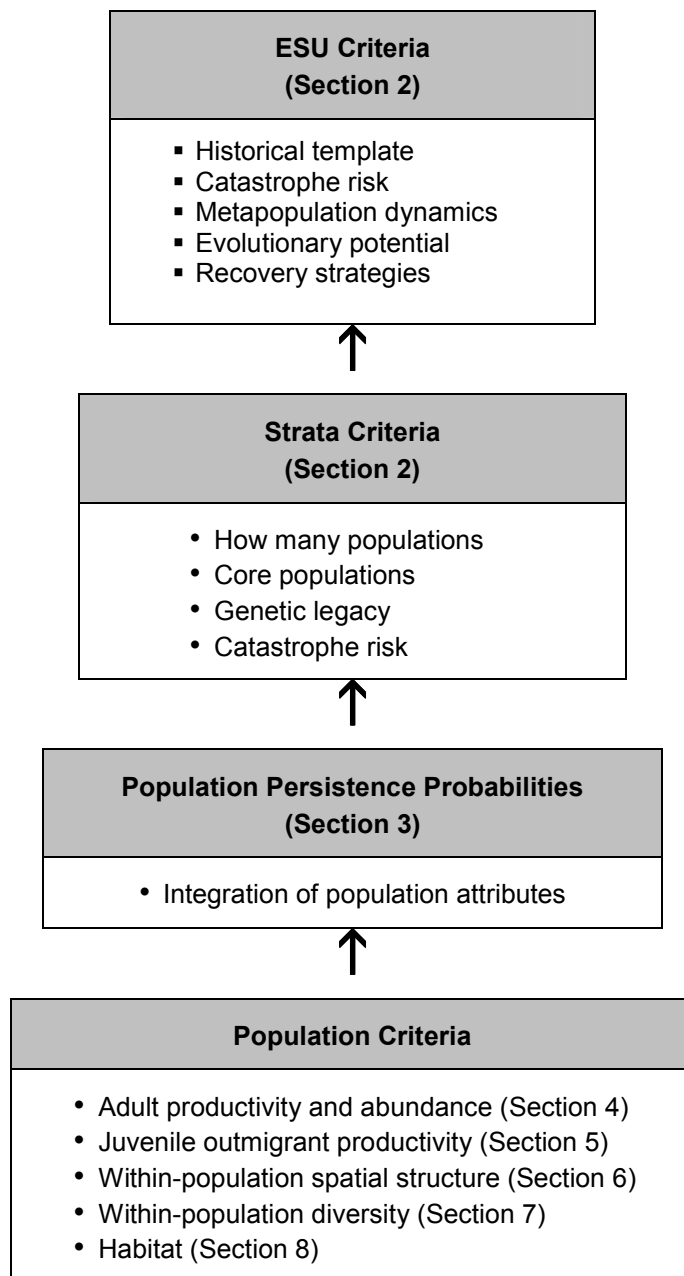


Figure 1.1 Approach to ESU criteria. The bullets list key considerations involved in each criterion. The section numbers refer to the section of this document that addresses each issue.

- The population is subject to extinction from catastrophic events.
- The population is experiencing genetic degradation.
- The habitat is experiencing progressive degradation.

We considered each factor but worked to develop criteria that could define viable populations and ESUs without regard to which factors are causing a population to be at risk. We did not attempt to determine the primary risk factor currently acting on any particular population or ESU.

The extinction risk factors can be subdivided into specific factors that have led to population and ESU decline. For example, if a population is at risk because density-independent survival is too low, the proximate cause may be poor water quality, excessive harvest, high hydrosystem passage mortality, or predation by exotic species, to name a few possibilities. In general, the salmon viability criteria do not attempt to partition the sources of mortality. For example, adult productivity and abundance criteria examine the spawning population after the fish have experienced all sources of mortality. If a population is currently at risk of extinction because survival is too low, the criteria allow us to evaluate whether survival has improved; however, the criteria do not presuppose what needs to be fixed to improve survival. In this respect, the productivity and abundance criteria may be referred to as “mortality-source neutral.” To address a specific factor of concern, the criteria may be considered “harvest neutral” in that the criteria do not stipulate a specific harvest level. In combined consideration of all mortality sources, any harvest strategy that allows the criteria to be achieved would be considered possible in order to achieve a viable status.

Although the salmon attribute criteria are not intended to identify specific actions required for recovery, the ability to achieve different criteria will clearly be affected differently by different potential factors for decline. For example, the spatial structure and diversity criteria within a population are heavily affected by habitat structure, and the criteria described below reflect this fact. Again, the criteria do not specifically address what factors need to be addressed in any particular population.

Some of the population attribute criteria described in this report—notably the spatial structure, diversity, and habitat criteria—will require additional analysis at finer spatial scales than have been undertaken by the WLC-TRT so far. The criteria on these topics in this report essentially amount to guidelines; population-specific criteria will need to be developed so that concrete goals can be established.

2. ESU-LEVEL VIABILITY

ESU-LEVEL CRITERIA GUIDELINES
1. Every stratum (life history and ecological zone combination) that historically existed should have a high probability of persistence.

STRATA CRITERIA GUIDELINES
1. Individual populations within a stratum should have persistence probabilities consistent with a high probability of strata persistence.
2. Within a stratum, the populations restored/maintained at viable status or above should be selected to: <ul style="list-style-type: none"> a. Allow for normative metapopulation processes, including the viability of “core” populations, which are defined as the historically most productive populations. b. Allow for normative evolutionary processes, including the retention of the genetic diversity represented in relatively unmodified historical gene pools. c. Minimize susceptibility to catastrophic events.

ESU-LEVEL RECOVERY STRATEGY CRITERIA GUIDELINES
1. Until all ESU viability criteria have been achieved, no population should be allowed to deteriorate in its probability of persistence.
2. High levels of recovery should be attempted in more populations than identified in the strata viability criteria because not all attempts will be successful.

Overview

As the unit listed under the ESA, the ESU is also the unit that must be considered for delisting. Part of the process for developing ESU delisting criteria involves describing the biological attributes of a viable ESU, which is defined as one with a high probability of persistence. The persistence probability is the complement of the extinction risk (i.e., persistence probability = 1 – extinction probability), and both terms are used in this document. The ESU viability criteria proposed in this report provide some flexibility in deciding which populations need to be restored to what status. Finalization of a viable ESU scenario will require policy/technical interaction.

To develop the viability criteria we applied the VSP approach outlined in McElhany et al. (2000). The basic strategy is to identify historically independent populations, develop criteria

describing viable independent populations (i.e., VSPs), then determine how many and which populations need to be at a particular status relative to the VSP criteria for the ESU as a whole to have an acceptably low extinction risk. Elsewhere in this document we describe the population-level viability criteria for the attributes of productivity, abundance, juvenile outmigrants, diversity, habitat, and spatial structure. In this section, we address the question of how many and which populations need to be in what viability status.

Clearly the most precautionary approach—the one that would give the highest probability of ESU persistence—would be for all historical populations to meet or exceed the viable population criteria. However, several ESUs historically contained a relatively large number of populations (Table 2.1) and it is possible that a subset of the historical populations can provide an adequate probability of ESU persistence. For example, the Lower Columbia chinook ESU is estimated to have consisted of 31 demographically independent populations. As supported by intuition and simple probability modeling (below), the probability of extinction for an ESU with 30 of the 31 historical populations in viable condition is not likely to be much different than the probability of extinction for a population with all 31 historical populations in a viable condition. In either case, the probability is low. If we allow that not all historical populations need to be viable for the ESU to be viable, we are confronted with the questions exactly how many are needed and does it matter which ones.

McElhany et al. (2000) provides seven guidelines for determining how many and which populations are needed for a viable ESU.

1. The ESU should contain multiple populations.
2. Some populations within the ESU should be geographically widespread.
3. Some populations should be geographically close to each other.
4. Populations should not all share the same catastrophic risk.
5. Populations that display diverse life histories and phenotypes should be maintained.
6. Some populations should exceed VSP guidelines.
7. Evaluations should take into account uncertainty about the ESU-level process.

These guidelines are motivated primarily by concern about catastrophic risks, metapopulation dynamics, and long-term evolutionary dynamics. If an ESU contained only a single population, there is a possibility that it could be driven extinct by a single catastrophic event. This necessitates multiple viable populations within a viable ESU. The risk of ESU extinction from

Table 2.1 Estimated number of historical demographically independent populations.

ESU	Historical Populations
Lower Columbia chinook	31
Lower Columbia steelhead	23
Columbia River chum	16
Upper Willamette chinook	7
Upper Willamette steelhead	4

Source: Myers et al. (2002)

catastrophic events can be further reduced by careful consideration of which populations are restored or maintained at viable status. Appendix K is an exploration of the spatial distribution and frequencies of potential catastrophic events affecting Pacific salmonid populations in the WLC. For reasons discussed in the appendix, it is difficult to predict catastrophic risks to salmon populations, but several general conclusions relevant to setting viability criteria are possible.

- Extinction risk is reduced if viable populations are spatially distributed throughout the ESU.
- Populations that utilize different types of environments experience different catastrophic risks. Having populations in different environments reduces the likelihood that a single catastrophic event would affect every population in an ESU.
- Because of the spatio-temporal patterns of catastrophic events, fish with different life histories that share the same river basin may be affected differentially by the same catastrophic event.

As discussed in the section on within-population diversity, genetic and life-history diversity helps buffer a population from extinction. Different genotypes and life histories are likely to be favored under different environmental conditions, and, as a consequence, diverse populations have a higher probability of persistence. As at the population level, diversity at the ESU level can increase the persistence probability of the ESU. If an ESU contains populations with different genetic or life-history types, it is less likely to go extinct because not all populations would respond to the environment in the same way. Conditions that cause one population to decrease in abundance may not affect another population at all, or may actually cause it to increase. In fact, this buffering effect can occur simply as a result of spatial diversity, in which different populations respond differently to conditions as result of their spatial dispersal.

Restoring and maintaining populations with different genetic and life-history types is also important for maintaining the evolutionary processes that are a part of any functioning biological system. The environment will change in the future and the existence of genetic diversity is essential if the ESU is to respond evolutionarily to that change and persist. The ESU concept is based on the premise of protecting the “evolutionary legacy” of an ESU (Waples 1991), and part of the evolutionary legacy of an ESU is the diversity within it. The concept of an evolutionary response to change is particularly relevant in the face of directional environmental change caused by humans. Some of the most significant long-term changes to affect salmon could be effects of global climate change as a result of the “greenhouse effect.” It is difficult to predict with confidence which genotypes or life-history types will be favored under any future changes in the environment. Thus to ensure that at least some genotypes exist that will be favored in the future, the ESU-level criteria incorporate diversity considerations.

Establishing ESU-level viability criteria is hampered by our limited understanding of many of the processes that operate at the ESU scale. For example, ESUs may operate as a metapopulation at large spatial and temporal scales, with populations naturally experiencing periodic extinction and recolonization (Levins 1969, Hanski and Gilpin 1997). This suggests that metapopulation modeling could be used to help determine ESU-level viability criteria. However, accurate estimates of the key parameters needed for quantitative metapopulation modeling, such as dispersal rate and local extinction probabilities, are simply not available. Although the concepts of metapopulation theory informed our thinking about ESU-level criteria, we conducted only very limited quantitative modeling (see “Number of Populations per Stratum,” page 11).

Rather than rely on quantitative modeling, we focused on the principles suggested by considering the importance of catastrophic events and among-population diversity to develop the general framework of the viability criteria. We also relied on the general concept that the historical ESU was viable and that it provides the only known template of a functioning ESU. As noted above, it may not be necessary for every historical population in an ESU to be at viable status for the ESU to be viable. However, confidence in ESU viability can be enhanced if the populations restored and maintained at VSP status recreate the basic structure of the historical template. Reference to the historical template motivates several components of the ESU-level viability criteria.

In addition to the biological concerns about ESU persistence, answering the questions how many and which populations are needed to delist an ESU must be informed by the language of the ESA itself. The ESA states that a species may be listed if it is threatened or endangered in "... all or a significant portion of the range." Since the ESU is functionally considered a "species" for the purposes of the ESA, presumably an ESU could only be delisted if it is no longer at risk of extinction in a significant portion of its range. The word significant is not defined in the ESA, and it is not clear whether "a significant portion of the range" would constitute more of the range than is required for species viability. In this document, we focus on viability and do not rely on the "significant portion of the range" language of the ESA in developing criteria.

Viability criteria describe a set of conditions, which, when met, would indicate that a population or ESU has a high probability of persistence. We have generally focused on the desired future conditions and have not discussed actions or strategies for reaching these goals in developing the viability criteria in this document. However, we did consider it important to include reference to two "recovery strategy guidelines" regarding the ESU viability criteria. These guidelines are included because presenting the viability criteria alone could suggest recovery strategies that would be inconsistent with actually reaching the criteria goals.

Viability Criteria Approach

In considering all the concepts discussed above to develop the viability criteria, the TRT partitioned the populations in an ESU into a number of different strata, then specified a risk evaluation system for deciding how many populations within each stratum should be at what status (Figure 1.1). The strata are defined based on two factors: (1) major life-history differences and (2) ecological zones (Table 2.2). The partitioning based on ecological zones also results in a partitioning based on spatial distribution. If the ESU contains populations in each stratum, it will have a relatively low extinction risk from catastrophic events, correlated environments, and loss of diversity. In addition, the ESU will have some semblance of its historical structure, which increases confidence in ESU viability. Attributes of the different strata are described briefly below and in more detail in Appendix A.

Table 2.2 Estimated number of populations in different strata in the Willamette/Lower Columbia domain.

ESU	Ecological Zone ^a	Run Timing ^a	Historical Populations ^b
Lower Columbia chinook	Coast Range	Fall	7
	Cascade	Fall	9
		Late fall	2
		Spring	7
	Columbia Gorge	Fall	4
		Spring	2
Lower Columbia steelhead	Cascade	Summer	4
		Winter	14
	Columbia Gorge	Summer	2
		Winter	3
Columbia chum	Coast Range	Fall	7
	Cascade	Fall	7
	Columbia Gorge	Fall	2
Upper Willamette chinook	Willamette	Spring	7
Upper Willamette steelhead	Willamette	Winter	4
Total			81

^a Each run timing and ecological zone combination is a separate stratum.

^b The historical number of populations is based on Myers et al. (2002).

The life-history factors defining the strata are based on major differences that are the basis of some population designations (Myers et al. 2002). Lower Columbia chinook salmon are partitioned into spring, fall, and late-fall runs. Lower Columbia steelhead are partitioned into summer and winter runs. Populations of the three other listed ESUs in the WLC domain, Columbia chum salmon, Upper Willamette chinook salmon, and Upper Willamette steelhead, each consist of a single major life-history type and were not partitioned by run timing. While the different life-history types are named by run timing, each run type exhibits a number of different, presumably coadapted, life-history characteristics. Differences between the Lower Columbia chinook and steelhead life-history types are described in some detail in the TRT document identifying populations in the WLC domain (Myers et al. 2002). Loss of major life-history types was considered significant during decisions to list Lower Columbia River ESUs under the ESA. Myers et al. (1998) indicated that the Lower Columbia River chinook salmon ESU was listed, in part, because the biological review team (BRT) was unable to identify a single healthy spring-run chinook salmon population. Busby et al. (1996) indicated that one of the reasons the BRT listed Lower Columbia River steelhead as threatened was concern over the status of summer steelhead in this ESU.

Ensuring that populations persist in each ecological zone reduces risk of ESU extinction from catastrophic events and loss of diversity. Ecoregions defined by the U.S. Environmental Protection Agency (EPA) (Omernik 1987) were used to help define ecological zones for the WLC ESUs. The EPA ecoregions were designated for the contiguous United States based on

soil, topography, climate, potential vegetation, and land use (see Appendix B). Hughes et al. (1987) noted a strong link between ecoregions and freshwater fish assemblages. Salmon and steelhead populations in the Lower Columbia River ESU primarily cover the Coast Range, Cascade Range, and Columbia Gorge (East Cascades) ecoregions (Myers et al. 1998, Busby et al. 1996, and Johnson et al. 1997). Spring-run chinook salmon and winter steelhead in the Willamette ESU occupy the Cascade and Willamette Valley ecoregions (Busby et al. 1996, Myers et al. 1998). The EPA ecoregions were modified slightly to create more salmon-appropriate “ecological zones” (Appendix A). Because the climate, geology, and ecological processes in each ecological zone are different, it is expected that different ecological zones are unlikely to be affected by the same catastrophic event (Appendix E). The ecological zones represent distinct selective environments, and the persistence of populations in each zone is expected to preserve much of the ESU’s diversity.

Number of Populations per Stratum

Given the value in restoring and maintaining viable populations within each stratum, we need to decide how many and which populations should be at what status within a stratum. This section addresses “How many?” and the next section addresses “Which ones?” It is important to note that we are not striving for a zero extinction risk for each stratum. The unit listed under the ESA, the focus of the viability criteria described in this section, is the ESU. However, ESU viability is more likely if each stratum has a relatively low probability of extinction.

As one approach to considering how many populations are needed per stratum, we estimated the probability that there would be no populations remaining in a stratum after some period of time, given an initial number of populations and an independent, identical, per-population extinction rate. Under these assumptions, the stratum extinction risk declines exponentially with the initial number of populations as

$$\phi = \theta^\eta$$

where

ϕ is the probability that all the populations in a stratum will be extinct within y years,

θ is the probability that a single population will go extinct in y years, and

η is the number of initial populations in the stratum.

The probability of stratum extinction for a number of different per-population extinction risks when $y = 100$ years is shown in Figure 2.1. To really parameterize this equation, we would need to know the per-population extinction risk. This is not something we can estimate with any degree of precision. Meeting all population-level viability criteria is likely to result in a low probability of extinction from processes internal to the population, but we cannot calculate exactly how low. In addition, populations are subject to extinction from external catastrophic events, and for reasons discussed in Appendix K, it is difficult to quantify the per-population catastrophe risk. The calculation also makes the critical assumption that population extinction risks are independent. This is most certainly not the case, because correlated environments and catastrophic risks increase the likelihood that multiple populations can go extinct at the same time. The probability calculations indicate that, in general, having 2 to 3 populations with a low extinction risk in a stratum provides a relatively significant reduction in risk compared to a single

population, but having four or more populations does not greatly reduce the risk. There is a continuous exponential decline in extinction risk as more populations are added; however, the practical increase in risk reduction shows diminishing returns. Because the simple model is likely to underestimate extinction risk, we concluded that it is potentially useful in defining a lower bound on the minimal number of populations, but not informative beyond that limited application. Based on this simple probability analysis and professional judgment about the point of diminishing returns, the TRT concluded that a viable ESU should contain at least two viable populations per strata, but that additional considerations are needed to estimate how many more than two are required.

We approached the stratum risk criteria from the perspective of the historical template. The approach is based on the principle that the historical population structure of the strata produced a relatively low risk of extinction, and the closer the population structure is to that historical structure, the lower its extinction risk. Conversely a population structure that deviates greatly from the historical structure would be considered at high risk. Although this general argument is logically sound, it does not allow the identification of a “bright line” viability threshold for stratum criteria. How close to historical is good enough? Below, we present a stratum evaluation system with some suggested benchmarks for criteria. However, it must be recognized that these benchmarks are based on professional judgment of high, moderate, and low risk categories.

Another limitation of the simple stratum risk calculation presented above is the

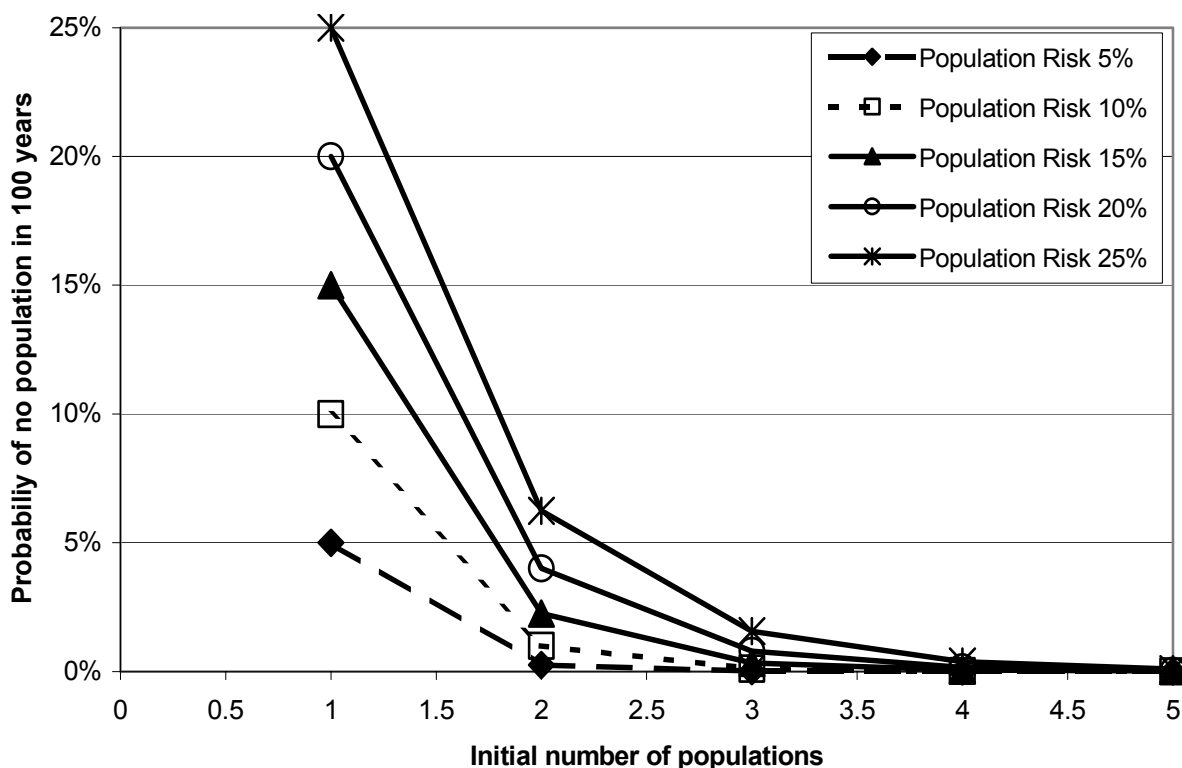


Figure 2.1 Probability of losing all the populations in a stratum within 100 years as a function of the initial number of populations, assuming populations are independent. Each curve represents a different per-population probability of extinction in 100 years.

assumption that all populations will have the same extinction risk. In fact, population extinction risks vary by population and a viable ESU scenario is likely to contain populations at different risk levels. This complication makes development of a quantitative metapopulation model that predicts viable ESU scenarios even more challenging. In a previous draft of this document (May 2002), we developed stratum criteria based primarily on consideration of a single risk level, that of a viable population as defined in McElhany et al. 2000 (i.e., negligible risk of extinction in 100 years). In the earlier draft, we specified the number of populations needed to meet or exceed VSP status. A second, higher risk level was also discussed (i.e., that associated with an effective population size of 500 spawners). This earlier approach failed to consider the entire range of potential population risks in a viable ESU scenario. Some populations may have a much lower risk of extinction than that defined in McElhany et al. 2000, and others may have a much higher risk. Because of issues related to population connectivity and metapopulation dynamics, populations at all risk levels have the potential to contribute to ESU viability, and the challenge is to identify the combination of populations at different risk levels that leads to a viable ESU.

Although population persistence probability is a continuum from near 100% (at least in 100-year time frames), to near 0% (or even already extirpated), there is limited precision in persistence probability estimates. Thus, to develop ESU-level criteria, we have divided the continuum into five categories (Table 2.3). Population risk assignment is limited to five categories because a continuous scale could impart a false sense of precision regarding the estimates. With some hesitation, we provide quantitative persistence probabilities associated with each qualitative persistence category in Table 2.3. Simply stating the quantitative thresholds implies that persistence can be measured with some degree of precision. Again, we do not believe it possible to accurately estimate persistence probability. However, it is possible to provide rough quantitative estimates of persistence probabilities associated with population productivity and abundance (Section 4), and Table 2.3 aids in associating such estimates with the persistence categories. The majority of sections in this document (Sections 2–8) address the establishment of criteria for assigning populations to risk categories. In Section 3 (“Integrating Population Risk”), we present an approach for combining information on individual population attributes (productivity, abundance, diversity, habitat, and spatial structure) into one of these five risk categories.

Using the categories described in Table 2.3, we have developed an approach for describing how many populations need to be at what status in each stratum. The approach uses

Table 2.3 Description of population persistence categories.

Population Persistence Category	Probability of Population Persistence in 100 Years	Description
0	0–40%	Either extinct or very high risk of extinction.
1	40–75%	Relatively high risk of extinction in 100 years.
2	75–95%	Moderate risk of extinction in 100 years.
3	95–99%	Low (“negligible”) risk of extinction in 100 years (viable salmonid population).
4	>99%	Very low risk of extinction in 100 years.

the average risk category of the populations in the stratum. The average is based on the historical number of populations, not the current number (e.g., if an historical population is extirpated, it is not ignored but is entered into the average as a 0.) Taking the metric of stratum extinction risk as an average of the individual populations risks allows the stratum metric to be scaled to the historical number of populations (i.e., strata that historically contained more populations would need more populations in a low-risk category than strata with few historical populations). This will result in viable ESU scenarios that resemble the historical population structure in terms of the number of populations and are consistent with the historical template concept. Using an average approach also recognizes that having some populations that exceed the VSP population criteria (i.e., category 4) can help mitigate the risk from populations with higher risk categories (i.e., categories 0–2). In examining stratum averages, we developed the general guidelines for stratum risk shown in Table 2.4. We considered, but did not provide, quantitative persistence probabilities associated with each stratum persistence category. We have no way of providing quantitative estimates of stratum persistence, and did not want to impart a false sense of precision.

The professional judgment for the thresholds was made after considering the averages of all the possible combinations of population values that could occur in a stratum. As a rough guide, the TRT identified averages that produced combinations considered functionally similar to the strata criteria developed in the previous draft of this document (i.e., the greater of 2 populations or 50% of the historical populations in a stratum should be at viable status or higher, and all extant populations should have an effective population size of at least 500). The new averaging approach provides more flexibility in defining viable ESU scenarios, but should describe similar levels of risk as those in the previous draft. Tables showing the averages of all possible combinations of populations are available on the WLC-TRT Web site at http://research.nwfsc.noaa.gov/cbd/trt/trt_wlc/viability_report.htm.

The average population risks in Table 2.4 are proposed as thresholds for the strata criteria guideline related to the number of populations.² The reliance on professional judgment for the

Table 2.4 Stratum persistence categories based on averages of individual population risks. The category thresholds are based on *professional judgment*.

Stratum Persistence Probability Category	Average of Population Risks
Low persistence	Average < 2
Moderate persistence	2 ≤ Average < 2.25; at least two populations ≥ 3
High persistence	Average ≥ 2.25; at least two populations ≥ 3

² Note on the mathematical properties of the average population risk: The strata average does not indicate the expected value of the persistence probability for populations in the strata. There is a nonlinear relationship between the population persistence probabilities and the population persistence categories. That is, some population persistence categories are associated with very wide ranges (e.g., category 1 has a range of 40%) and other population persistence categories are associated with more narrow ranges (e.g., category 3 has a range of 10%). The population average is used as a metric of stratum persistence probability, not as an expected value. The nonlinearity in the population categories does not reduce the utility of the metric.

establishment of these stratum average thresholds is in many ways unsatisfying. However, the inability to quantitatively model the relevant processes leads to a reliance on professional judgment for decisions about stratum risk.

Selection of Populations in a Stratum

Within a stratum, careful selection of the populations restored or maintained at a high persistence probability status (i.e., category 3 or 4) can increase the probability of ESU persistence. Within a stratum, the populations restored or maintained at viable status should be selected so as to:

- Allow for normative metapopulation processes, including the viability of “core” populations, which are defined as the historically most productive populations.
- Allow for normative evolutionary processes, including the retention of the genetic diversity represented in relatively unmodified historical gene pools.
- Minimize susceptibility to catastrophic events.

Metapopulation Processes

A metapopulation is a group of relatively independent populations that interact through the movement of individuals among them. Individual populations within the metapopulation may be extirpated (or nearly so) by internal or external processes and subsequently be recolonized (or “rescued”) by migrants from neighboring populations. In a stable metapopulation, the natural rate of population extirpation is matched by the natural rate of recolonization (see McElhany et al. 2000 for discussion of metapopulations and Pacific salmonids). On relatively large temporal and spatial scales, the populations in an ESU are expected to act as a metapopulation, within which the exchange of migrants among populations has an important impact on the ESU’s long-term persistence. Section 6 (“Within-Population Spatial Structure Criteria”) discusses these issues at smaller spatial and temporal scales.

Simply having enough populations is one important consideration for metapopulation persistence. The stratum average approach discussed above is an effort to identify an adequate number of populations as a function of the historical number. In addition to sheer numbers, other issues may be important, such as the distribution of populations and the particular dynamics involved. Source-sink dynamics are one type metapopulation process in which some populations are consistently more productive than others. These source populations can serve as a source of migrants to recolonize neighboring, less productive (sink) populations that are periodically extirpated or depressed in abundance. A key strategy in conserving a source-sink metapopulation is to protect the source population (McElhany et al. 2000).

Under historical conditions, not all salmon populations had the same productivity. Myers et al. (1998) demonstrated that some salmon populations were more productive than others. Using a life-cycle model for coho salmon, Nicholson and Lawson (2000) indicated that in periods of low ocean productivity only salmon inhabiting the best rivers or reaches survived. Population abundance is not the same as productivity, but variation in abundance or density

among populations provides some indication of variability in productivity. As an example of this variability, the lower Cowlitz River fall chinook salmon population is estimated to have historically consisted of 54,000 fall chinook spawners in a 441-square-mile drainage area, giving a density of 122 spawners per square mile. In contrast, there were an estimated 5,000 fall chinook spawners historically in the 98-square-mile Coweeman River drainage, yielding a density of 51 spawners per square mile (Appendix J). Historical population estimates for summer steelhead also indicate similar levels of among-population variability in abundance, with the East Fork Lewis River spawner abundance estimated to be 400 fish, while the Wind River spawner abundance was estimated at 2,300.

We define the historical salmon populations that were the center of productivity and abundance for a stratum as core populations. Note that this definition of core potentially differs from other uses of the term in conservation biology and natural resource management. Some researchers and managers have used the term core to describe current population strongholds or to identify areas for intensive restoration or protection. Our definition is based on historical fish performance and may or may not correspond to current status or management strategies. Based on the historical ESU template concept, having at least some core populations with a high probability of persistence is likely to provide the highest probabilities for ESU persistence. The recovery of core populations is likely to create an ESU with a strong resemblance to the historical structure. The basis of the historical template concept is that the historical ESU was viable, and the more an ESU resembles the historical structure, the more confidence we have that it will be viable. In Appendix B, we identify core populations in the WLC domain.

Evolutionary Processes

As discussed in more detail in Section 7, the genetic variability within a salmon population allows salmon to adapt to a changing environment. Given that genetic traits underlie the productivity and ecological potential of a population, conservation biologists have placed a high priority on protecting this diversity because it is a key to species survival. In fact, the intent of the ESA is to protect the ESU, which is defined as representing “an important component in the evolutionary legacy of the species.” Human transfer of salmon between basins has been widespread for more than 100 years, and many of the current populations differ genetically from the historical populations, resulting in a loss of diversity. In deciding which populations to restore and maintain at viable status it is important to include populations that still represent the historical diversity. Appendix B of this document and Appendix C of Myers et al. (2002) describe the genetic and life-history relationships between current and historical populations; the information in these appendices can help identify current pools of diversity. Maintaining these pools of diversity should be a key consideration when populations are ranked or prioritized to achieve viability goals.

Catastrophic Risk

The presence of viable populations in each stratum is expected to substantially reduce the risk of extinction of the ESU from catastrophic events. Careful selection of populations within a stratum can further reduce the extinction risk due to catastrophic events. For example, the Lower

Columbia River ESUs will be less vulnerable to catastrophic loss if viable populations exist on both sides of the Columbia River. As another example, multiple salmon populations could be impacted by the same volcanic event, and the existence of viable populations in watersheds that are not entirely located on the same volcano can increase ESU viability. Appendix L describes the spatial distribution of some potential catastrophic events that could affect salmon populations. This information can be used for ranking or prioritizing which populations should be restored and maintained at viable status. In an effort to minimize risk from catastrophic events, the populations within a stratum will generally be geographically widespread.

Approach to Selecting Populations

From a biological perspective, determining which populations in a stratum should be restored and maintained at viable status requires simultaneous consideration of metapopulation processes, evolutionary processes and catastrophic risk. Because of the many contingencies involved, we recommend that the evaluation of proposed stratum viability scenarios be conducted using professional judgment. The appendices listing core populations and genetic legacy populations provide guidance on the selection of populations for viable stratum scenarios. However, it may not be necessary for all of the core populations and genetic legacy populations to be viable for the stratum and ESU to be viable. Determining exactly whether a particular population is needed for a functioning metapopulation or to reduce risk from catastrophic events depends on which other populations are at viable status. Since the number of potential combinations of populations at viable status is potentially very large, we did not find it feasible to develop a simple mathematical algorithm for determining which populations to select and instead rely on professional evaluation based on the relevant biological principles.

ESU-Level Viability Criteria and Strata Persistence

A precautionary approach to ESU viability would require all strata to have a high probability of persistence. A less precautionary approach might consider a mixture of strata with high and moderate persistence probabilities. We suggest that a viable ESU should have all strata in the high-persistence category.

How precautionary to be in setting delisting criteria at the ESU scale is ultimately a policy decision. However, the appropriate attribute threshold (e.g., stratum average) associated with each persistence category is a scientific question. Unfortunately, it is not a question that can be answered with precision, and there is ample room for scientific debate. For several reasons, we have not attempted to associate qualitative descriptions of ESU persistence (e.g., high, low) with quantitative thresholds (e.g., a high persistence is a 99% probability in 100 years). Such associations rely on societal and policy perceptions of high and low and the relation of these perceptions to the ESA. In addition, scientific estimates of persistence probability at the ESU scale are even less precise than at the population and stratum scales. The imprecision associated with each assessment level (population attribute → population summary → stratum → ESU) is propagated up to the ESU level to create a very uncertain estimate. If policy makers were to supply an explicit acceptable probability of ESU persistence, we could provide a professional judgment estimate of criteria thresholds. However such estimates would be extremely imprecise,

thus relying on qualitative advice, such as describing potential ESU scenarios as “more precautionary” or “high risk,” may be the best the TRT can provide without implying greater precision than actually exists.

Recovery Strategy Criteria

The viability criteria in this document describe scenarios that, if observed, would indicate a population or ESU has a high probability of persistence. These criteria should be clearly distinguished from a strategy for how to actually recover the ESU. The viability criteria describe a future desired state. Given that some efforts to recover populations will inevitably not be successful, a prudent recovery strategy would require attempting to recover more populations to a higher status than is stipulated simply by the viability criteria. As a simple example, imagine that the target viability criteria required that three populations in a stratum be at VSP status (i.e., category 3). If there is an 80% chance that any given population recovery effort will be successful, there is only a 51% probability that three populations will be recovered if recovery is only attempted in three populations (Table 2.5). In this example, to have a greater than 95% probability of achieving the target of three populations, recovery would need to be attempted in at least six populations.

In this document, we do not assess the probability that any given recovery strategy will be successful. This probability depends on the recovery strategy selected. Information on the likelihood of success for different recovery strategies will need to be developed as action plans are created. However, no population recovery strategy is guaranteed to be successful, and it will be important to “overshoot” the number of populations in which recovery is attempted.

Another issue related to the distinction between viability criteria and a recovery strategy involves the protection of extant populations. Viable ESU criteria may allow for extirpated populations. Although this may describe an acceptable end state, it would be highly risky to allow current populations to decline in the short term. As noted in the previous paragraph, it is important as a recovery strategy to overshoot the goal, and striving for an exact target is likely to fail. As a practical matter, high recovery levels likely will need to be attempted in most, if not all, current natural production areas. Recovery strategies should consider the fact that we do not know *a priori* which population recovery attempts will be successful. There is a particular danger in writing off any extant populations, which might permanently remove options for ESU

Table 2.5 Probability of achieving recovery of at least three populations if the probability of a successful population recovery attempt is 80%. The probability of success for each population recovery attempt is considered independent.

Number of Populations in which Recovery Is Attempted	Probability of Recovering at Least Three Populations
3	51%
4	82%
5	94%
6	98%

recovery. Until all ESU viability criteria are met with regard to all populations, no population should decline from its current status.

Examples of Viable ESU Scenarios

The best way to explore the ESU viability criteria is through examples. In this section, we work through two example ESU scenarios that are consistent with the proposed criteria. In order to examine the criteria at the extremes, one of the examples involves the ESU with the most populations and strata (i.e., Lower Columbia chinook salmon), and the other involves the ESU with the fewest populations and strata (i.e., Upper Willamette steelhead). It is important to emphasize that these are EXAMPLES and NOT RECOMMENDATIONS for viable ESU scenarios.

To generate these example scenarios, a random collection of population persistence categories was selected for each stratum, such that the stratum average was between 2.25 and 2.5, and the stratum contained at least two populations of category 3 or higher (Table 2.6). Restricting the collections to an average of 2.25 to 2.5 puts all strata just barely above the high-persistence threshold. Strata with a higher average would also be considered at high persistence, but the behavior of the criteria near the threshold is likely to be of most interest. Strata with only two populations are the exception. In order to have at least two populations of category 3 or greater, the minimum possible average is 3. Random selection of the collection of population categories was considered the best way to develop an EXAMPLE. In practical application, target population persistence categories will not be selected at random but instead will consider the feasibility of restoration and other policy issues.

Once the collection of population persistence categories was selected, we considered metapopulation processes (e.g., core population), the evolutionary processes (e.g., populations that represented the genetic legacy) and the issues of catastrophic risk to decide which populations would be most important to be category 3 or higher (Table 2.7). These decisions relied on the information in Appendix B (core populations) and Appendix K (catastrophic risk), and (from Myers et al. 2002) Appendix C (genetic legacy). The key information from those appendices are summarized in Table 2.7. The prioritization we developed is not necessarily the only possibility, but it is an EXAMPLE that is consistent with the criteria principles.

Table 2.6 Random collection of population persistence categories used for example viable ESU scenarios.

ESU	Stratum	Number of Populations	Random Collection of Population Persistence Categories ^a	Average Population Persistence Category
Lower Columbia chinook salmon	Coastal fall	7	4,4,4,3,2,0,0	2.43
	Cascade fall	9	4,3,3,3,2,2,2,2,1	2.44
	Cascade late fall	2	3,3	3.00
	Cascade spring	7	3,3,3,3,2,1,1	2.29
	Gorge fall	4	3,3,2,1	2.25
	Gorge spring	2	3,3	3.00
Upper Willamette steelhead	Willamette	4	3,3,2,2	2.50

^a The collections were selected to have a stratum average between 2.25 and 2.5 and to contain at least two populations of category 3 or higher. The two strata with only two historical populations are an exception, with a stratum average of 3.0.

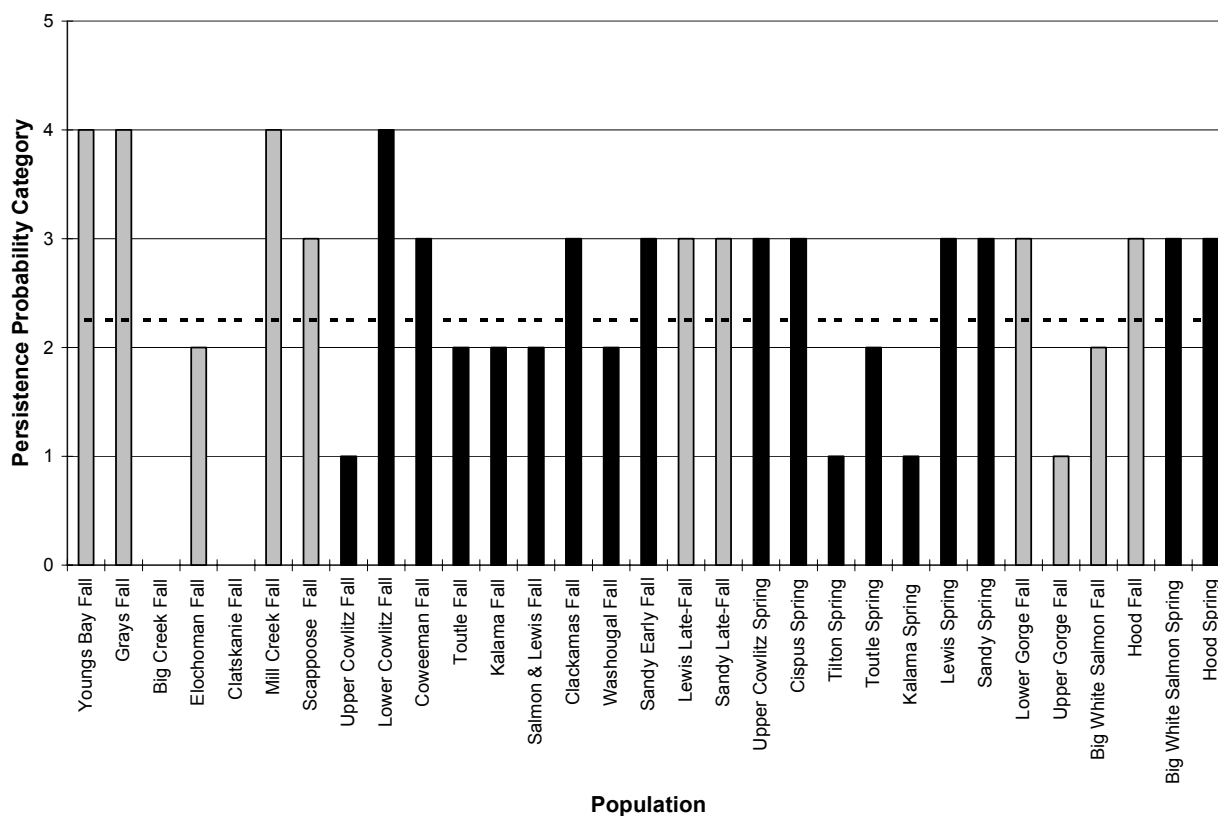


Figure 2.2 EXAMPLE Lower Columbia chinook salmon ESU scenario. The gray and black bars distinguish separate strata. Population risk categories were randomly selected from all of the strata combinations with an average of 2.25–2.5 (i.e., all strata are just above the low-risk threshold), except strata with only two populations. The dashed line indicates the 2.25 average persistence probability threshold for a high-persistence stratum.

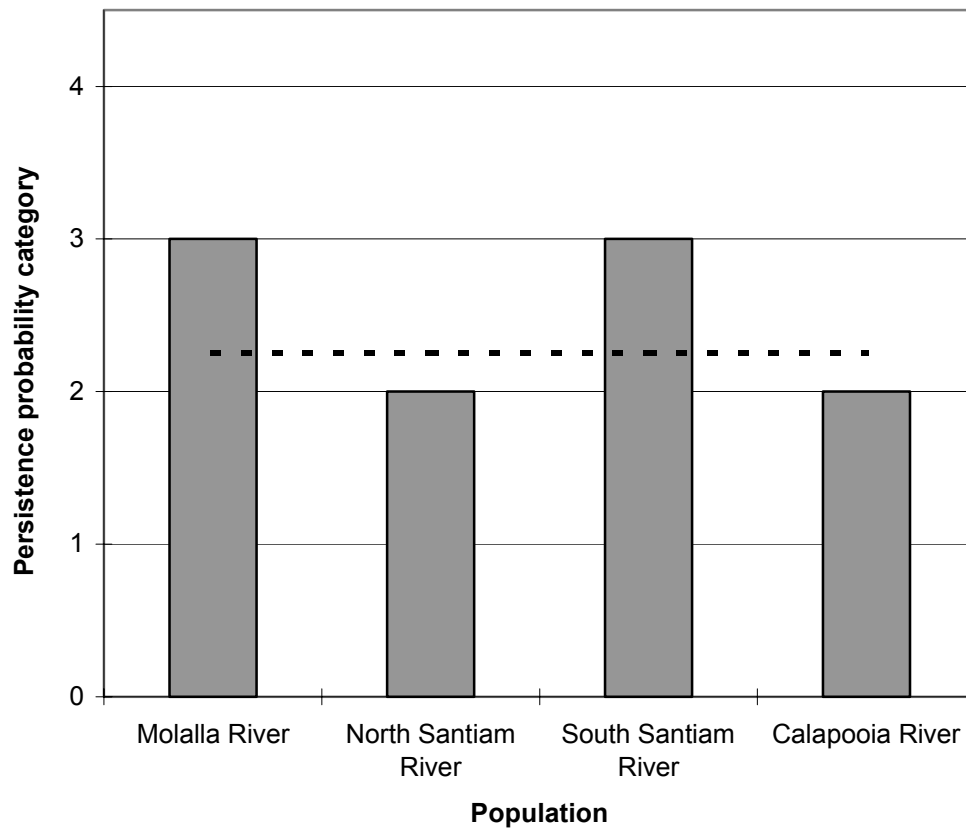


Figure 2.3 EXAMPLE Upper Willamette steelhead viable ESU scenario. This ESU consists of a single stratum. Population risk categories were randomly selected from all strata combinations with an average of 2.25–2.5 (i.e., all strata are just above the low-risk threshold). The dashed line indicates the 2.25 average persistence probability threshold for a high-persistence stratum.

Willamette/Lower Columbia Salmonid Viability Criteria

Table 2.7 EXAMPLE selection of populations needed above persistence category 3 for example viable ESU scenarios (continued on facing page).

ESU	Stratum	Core Populations	Genetic Legacy Populations
Lower Columbia chinook salmon	Coast fall	Elochoman Big Creek	
	Cascade fall	Lower Cowlitz Toutle Clackamas	Coweeman Salmon Creek-Lewis
	Cascade late fall	Lewis Sandy	Lewis Sandy
	Cascade spring	Upper Cowlitz Cispus Lewis Sandy	Upper Cowlitz Sandy
	Gorge fall	Lower gorge tributaries Upper gorge tributaries	
	Gorge spring	Big White Salmon River	
Upper Willamette steelhead	Willamette	North Santiam South Santiam	North Santiam South Santiam

Table 2.7 cont.

Catastrophic Risk Issues	Number of Populations in Random Collection of Category ≥ 3 (See Table 6)	Populations Selected To Be of Category ≥ 3
<ul style="list-style-type: none"> ▪ Earthquakes—low probability/high impact ▪ Landslides—from steep slopes, stream channels ▪ Disease from hatcheries—17.5 million hatchery fish raised and released in system ▪ Transportation oil spills—negligible to medium density 	4	<ul style="list-style-type: none"> ▪ Youngs ▪ Grays ▪ Mill ▪ Scappoose
<ul style="list-style-type: none"> ▪ Volcanoes—Mount St. Helens, Mt. Adams, Mt. Rainier, Mt. Hood ▪ Landslides—from volcanoes, steep slopes, and channels ▪ Disease from hatcheries—49.3 million hatchery fish raised and released in system ▪ Transportation oil spills—high density in urban areas 	4	<ul style="list-style-type: none"> ▪ Lower Cowlitz ▪ Coweeman ▪ Clackamas ▪ Sandy
<ul style="list-style-type: none"> ▪ Volcanoes—Mount St. Helens, Mt. Hood ▪ Landslides—from volcanoes, steep slopes, and channels ▪ Disease from hatcheries—10 million hatchery fish raised and released in system 	2	<ul style="list-style-type: none"> ▪ Lewis ▪ Sandy
<ul style="list-style-type: none"> ▪ Volcanoes—Mount St. Helens, Mt. Adams, Mt. Rainier, Mt. Hood ▪ Landslides—from volcanoes, steep slopes, and channels ▪ Disease from hatcheries—21.3 million hatchery fish raised and released in system ▪ Transportation oil spills—high density in urban areas 	4	<ul style="list-style-type: none"> ▪ Upper Cowlitz ▪ Cispus ▪ Lewis ▪ Sandy
<ul style="list-style-type: none"> ▪ Volcanoes—Mt. Adams, Mt. Hood ▪ Landslides—from volcanoes, steep slopes, and channels ▪ Disease from hatcheries—45.9 million hatchery fish raised and released 	2	<ul style="list-style-type: none"> ▪ Lower gorge ▪ Hood
<ul style="list-style-type: none"> ▪ Volcanoes—Mt. Adams, Mt. Hood ▪ Landslides—from volcanoes, steep slopes, and channels ▪ Disease from hatcheries—0.38 million hatchery fish raised and released 	2	<ul style="list-style-type: none"> ▪ Hood ▪ White Salmon
<ul style="list-style-type: none"> ▪ Landslides—from steep slopes, stream channels ▪ Disease from hatcheries—11.4 million hatchery fish raised and released ▪ Transportation oil spills—high density in urban areas 	2	<ul style="list-style-type: none"> ▪ South Santiam ▪ Mollala

3. INTEGRATING ATTRIBUTES AND ASSESSING POPULATION RISK OF EXTINCTION

Overview

The basic strategy for setting ESU viability criteria uses a hierarchical system to relate population scale attributes to ESU criteria (population attributes → population persistence probabilities → strata persistence probabilities → ESU criteria; Figure 1.1.) The population scale attributes are indicators of a population's extinction risk (or conversely, a population's persistence probability). McElhany et al. (2000) identified four important indicators of population extinction risk: productivity, abundance, spatial structure, and diversity. Working within the VSP framework as a starting point, the WLC-TRT developed the following general categories of indicators:

- adult productivity and abundance,
- juvenile out-migrant (JOM) productivity,
- population diversity,
- habitat, and
- spatial structure.

Subsequent sections of this document explore how each attribute can be used as an indicator of population extinction risk. In this section, we address the challenging task of integrating information on each attribute into an overall assessment of population extinction risk.

In Section 2, we introduced the approach of describing population persistence probability on a 0–4 qualitative scale, with 0 indicating a population with a low probability of persistence, and 4 indicating a population with high probability of persistence (Table 2.3). This 0–4 population scoring system is used as the basis for the ESU-level criteria. Because it is critical to this section, Table 2.3 is repeated here as Table 3.1.

Table 3.1 Description of population persistence categories.

Population Persistence Category	Probability of Population Persistence in 100 Years	Description
0	0–40%	Either extinct or very high risk of extinction
1	40—75%	Relatively high risk of extinction in 100 years.
2	75—95%	Moderate risk of extinction in 100 years.
3	95—99%	Low (negligible) risk of extinction in 100 years (VSP).
4	>99%	Very low risk of extinction in 100 years

Approach to Integrating Population Attributes

The proposed approach to integrating population attributes involves first evaluating the status of each population attribute separately on a 0–4 scale, then integrating the individual attribute values into an overall assessment of population status. The population attribute scores are based on the persistence category descriptions provided in each attribute section of this document, somewhat similar to those found in Table 3.2. For example, the population spatial structure would be evaluated based on whether it is consistent with a persistence probability that is high, low, or somewhere in between and assigned a 0–4 value accordingly. For some criteria (e.g., adult productivity and abundance and JOM growth rate), it may be possible to provide more quantitative thresholds associated with each level on the 0–4 scale. For other attributes (e.g., within-population diversity), it may not be possible to identify *a priori* quantitative thresholds, and more reliance on professional judgment will be required to determine the appropriate category. Issues related to the characterization of the individual attributes are discussed in the chapter on each attribute.

The TRT considered a number of possible procedures. Ideally, attribute persistence levels could be determined in a highly quantitative manner; however, in almost all cases the quantity and quality of available information necessary to derive such formulae were lacking (and will continue to be deficient under existing monitoring programs). Furthermore, the biological relationships among population characteristics are poorly understood. Data quality was a major concern for the TRT, and it was generally agreed that any population attribute measure needed to include some accounting for uncertainty due to poor data quality, in contrast to uncertainty due to environmental stochasticity. Furthermore, adjustments for poor data quality needed to be precautionary in nature and should be distinct from evaluations of the biological parameters.

A summary population profile table is a convenient way to view the status of populations in an ESU (Table 3.2). A table like this retains information on each individual attribute as well as the estimate of the overall population persistence category.

Table 3.2 Example “summary population profile table.”^a

Population	Population Attribute Persistence Categories					Population Persistence Category
	Growth & Abundance	JOM Growth	Spatial Structure	Diversity	Habitat	
A	3	1	1	1	1	2
B	4	ND ^b	3	3	3	4
C	0	0	0	0	0	0
D	2	1	ND	1	2	1
E	3	3	2	3	2	3
F	2	2	1	2	2	2

^a All values are completely made up and the individual attributes were arbitrarily integrated into an overall population persistence category.

^b ND = No Data. Indicates missing information; see Appendix C for TRT approach to missing data.

The TRT discussed a number of issues related to assigning values for each individual attribute and to assigning an overall persistence category for a population. Some of the key issues are as follows:

- *How much the procedure should rely on quantitative algorithms versus professional judgment.* Quantitative algorithms are potentially less subjective, but the majority of data will be qualitative, and professional judgment might provide the most accurate assessment of population status.
- *How to elicit professional judgment.* A number of procedures have been proposed for forming expert panels and eliciting professional opinion. It is important to capture diverse views within any expert panel.
- *How to incorporate uncertainty into the assessment.* There is uncertainty associated with each population attribute because of inherent variation in biological processes, scientific uncertainty about biological relationships, and uncertainty about data quality and measurement error. These different sources of uncertainty need to be explicitly identified and communicated during the risk assessment process.
- *How to handle attributes for which no information is available.* For some attributes, there may be no data available. For example, few locations currently have facilities to assess JOMs, and it is unlikely that JOMs can be assessed for every population. The assessment process needs to consider any additional risk associated with ignorance about a particular attribute.
- *How to account for the inherent correlation among all the population attributes.* All population attributes are expected to be correlated with one another to some extent. For example, the abundance of a population is correlated with its diversity, because processes like genetic drift are a function of population size. Habitat attributes are expected to be highly correlated with all the other attributes, particularly spatial structure (see Section 8). It is useful to consider each attribute separately because each one provides some independent information, but the correlations must be taken into account in weighting the value of each attribute into an integrated population persistence category.

In Appendix C we describe our approach to assigning persistence categories to populations. We intend to apply the method by assessing the current status of WLC populations. As the approach is applied, it may be modified as more is learned about the integration process.

4. POPULATION PRODUCTIVITY AND ABUNDANCE CRITERIA

ADULT POPULATION PRODUCTIVITY AND ABUNDANCE CRITERIA GUIDELINES

1. In general, viable populations should demonstrate a combination of population growth rate, productivity, and abundance that produces an acceptable probability of population persistence. Various approaches for evaluating population productivity and abundance combinations may be acceptable, but must meet reasonable standards of statistical rigor.
2. A population with a non-negative growth rate and an average abundance approximately equivalent to estimated historical average abundance should be considered to be in the highest persistence category. The estimate of historical abundance should be credible, the estimate of current abundance should be averaged over several generations, and the growth rate should be estimated with an adequate level of statistical confidence. This criterion takes precedence over criterion 1.

Overview

Key Issues

If a population experiences an unabated decline, it will eventually go extinct. This is true no matter how large the initial population or the cause of the decline. Thus, one of the primary metrics of population viability is an estimate of the long-term growth rate of the population. However, even if a population is not experiencing a long-term decline, there is some probability that it can go extinct. A population that is, on average, stable or increasing can go extinct as a result of stochastic (i.e., random) factors, which operate most strongly at small population sizes or as a result of catastrophic or other environmental events that may be independent of population size. The likelihood that a nondeclining population will go extinct is a function of the population's productivity.³ In these population productivity and abundance criteria, we focus on the processes that can lead to extinction of small populations. We address issues of size-independent catastrophic risk in the context of habitat criteria and in the context of ESU-level criteria.

The unit to which productivity and abundance criteria are applied can be very important. The appropriate unit for the criteria we have developed is a demographically independent population as described in McElhany et al. (2000). The demographically independent population concept is applied in the WLC domain in the draft TRT document identifying populations (Myers et al. 2002). The population units described in that document are used for the productivity and abundance criteria that follow.

³ The term intrinsic productivity refers the number of recruits per spawner that would occur at very low spawner abundance (i.e., if there were only a single pair of spawners). In this section, we use the more generic term productivity to refer to the general tendency of a population to return to dynamic equilibrium abundance if perturbed below that abundance. A population with high productivity would be considered resilient and have a relatively low risk of extinction.

As described below, a number of approaches can be used to set productivity and abundance criteria. Each approach is limited by its own critical assumptions and data requirements. The TRT recommends using the population change criteria (PCC) approach as a default method for setting productivity and abundance viability criteria. However, the default method involves only a general approximation of extinction risk and should be replaced with more detailed analyses when such analyses are supported by the data.

Approaches Considered

The TRT considered three basic approaches to estimating minimum population size. One approach relied on population viability analysis (PVA) modeling, in which minimum size thresholds were determined by estimating extinction risk as a function of the population size and other parameters. The other two approaches relied on estimation of historical abundance. The first of these was estimates of the historical population abundance based on broad-scale habitat-capacity analysis. The second historical approach was habitat productivity viability analysis (HPVA) modeling, as conducted by the Washington Department of Fish and Wildlife (WDFW) and the Northwest Indian Fisheries Commission (NWIFC). HPVA also uses habitat information, but makes productivity and capacity inferences based on fish-habitat relationships. For reasons explained below, we relied more on PVA than on the historical approaches in setting viability criteria.

In the PVA modeling, we focused on the role of environmental variation in identifying the minimum population size criteria. The risks confronting small populations include demographic stochasticity, environmental stochasticity, Allee effects, and genetic problems associated with inbreeding or the accumulation of deleterious mutations. Theoretical evidence suggests that in many cases the primary factor limiting the viability of small populations with nonnegative growth rates will be environmental stochasticity or catastrophes (Lande 1988 and 1993). Environmental stochasticity refers to the fluctuations in survival and fecundity associated with random environmental events. Even if a population is, on average, not declining, a chance sequence of bad years may drive a small population extinct, whereas a larger population would persist. Salmon are recognized as being highly variable in abundance, suggesting an important role for environmental stochasticity in setting minimum viable population sizes. Another key factor affecting extinction risk is a population's productivity or resilience, defined as its tendency to return toward equilibrium if pushed to low abundance.

The sections below and several appendices provide details on the PVA models, including the relationships between productivity, abundance, variability, and extinction. Specifically, we focus on two PVA-based analyses: the PCC approach and a two-life-stage recruit per spawner model. The PCC is considered the more precautionary of the two approaches.

We also evaluated to what extent historical abundance information could be used to develop viability criteria and to inform viability criteria developed by other methods. If we define historical as the time of pre-European settlement, most populations are assumed to have been viable at historical abundances and large enough to persist in the presence of natural environmental variability. Therefore, historical abundance could generally be used as a precautionary viability criterion. However, some populations, as defined in Myers et al. (2002), were relatively small, inhabited relatively unstable environments, and historically may have not have met the criteria of a viable population. For the majority of populations that were historically

viable, it is difficult to know whether historical abundance represents a minimum viability threshold. It is quite conceivable that a population below historical abundance levels would not be in danger of extinction. Allowing that populations below historical abundance may be viable, we explored the possibility of a viability criterion based on some specified fraction of historical abundance. The difficulty of this approach is that there is no clear way to link a fraction of historical abundance to viability except for the general (unquantifiable) statement that the closer a population is to historical abundance the more likely it is to be viable. It is also difficult to link historical abundance to viability, because simply looking at abundance does not provide information about resilience. Because there is no clear link between the fraction of historical abundance and viability, we relied primarily on the PVA modeling to obtain quantitative abundance viability criteria.

Estimates of historical abundance do, however, play an important role in the viability criteria by providing an upper bound on abundance criteria. As described in more detail below, a great deal of uncertainty is associated with the PVA modeling-based criteria. If the PVA modeling suggests a viable threshold abundance that exceeds the estimated historical abundance, the criteria use the historical abundance. This is because the thresholds established by PVA modeling (at least the population change criteria) are largely generic criteria applied to all populations in an ESU, and the circumstances in any given individual population may have allowed persistence at lower abundance than that suggested by the more generic standard. Although the PVA modeling threshold was considered generally appropriate, if credible historical analysis suggests that a population persisted at a lower abundance, the historical abundance was adopted as the viability criteria. Historical abundance was estimated based on a broad-scale habitat analysis (page 38 and Appendix I) and HPVA modeling (page 39 and Appendix J). Limited data based on historical surveys were also available for some populations, but in general these provided poor quantitative estimates of historical abundance (Myers et al. 2002).

The third approach considered in determining population abundance criteria was the HPVA modeling, as conducted by WDFW and the NWIFC and as described in Puget Sound TRT documents (PS-TRT 2002). HPVA is a specific application of the ecosystem diagnosis and treatment (EDT) method (Lichatowich et al. 1995, Mobrand et al. 1997) currently used in recovery planning. It uses up to 45 habitat and landscape attributes to predict the abundance, productivity, spatial distribution, and diversity of a population under particular habitat conditions. This approach does not identify a population viability threshold, but rather describes fish population attributes that would be expected under given habitat conditions. Because this analysis is not clearly related to extinction risk, we did not use HPVA analysis to establish viability criteria except as it informs the estimates of historical abundance. The EDT model can be evaluated under estimated historical habitat conditions to provide an estimate of historical population abundance. Although EDT documentation cautions against using the model as a predictive tool, HPVA results can produce qualitative information on historical abundance. HPVA for the WLC domain are provided in Appendix J. Although of limited application regarding productivity and abundance viability criteria, the HPVA analysis may be useful for developing other recovery planning goals. For example, HPVA information is being used to develop recovery targets for Puget Sound salmon species (Shared Strategy 2002).

PVA Modeling

Overview

This section provides a brief description of the PVA modeling approaches for setting productivity and abundance criteria. For a more complete explanation of conceptual underpinnings, assumptions, parameter estimation techniques, caveats, and references see Appendices D-H on PVA models.

A population with an unabated long-term decline will eventually go extinct. Thus, an intuitive viability threshold is the point at which a population replaces itself every generation and there is no long-term decline. This intuition is supported by PVA models, which indicate that long-term growth rate is one of the most informative predictors of population extinction risk. The median annual growth rate of a population, λ , can be estimated from an abundance time series as:

$$\hat{\lambda} = e^{\mu},$$

$$\mu = \text{mean} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right)$$

where N_t is the population abundance at time t . The critical value occurs where λ equals 1. If λ remains less than 1, the population will eventually go extinct. If λ remains greater than 1, the population will increase. Since the growth rate and minimum-size criteria are intertwined, they are discussed together in the material below.

Even if a population is, on average, not declining, there is still some probability that it will go extinct because of chance fluctuations in population abundance. The environment is variable, and a chance sequence of bad years may drive a population, particularly a small population, to extinction. The probability that a population “bounces” to extinction because of environmental variation depends on the size of the population, the amount of variation, and the population’s resilience, that is, its tendency to return toward an equilibrium value if pushed to low abundance. Resilience can be estimated as the intrinsic productivity of the population: intrinsic productivity is defined as the number of returning fish per spawner that would be produced if the population were at very low abundance. All else being equal, a population with higher intrinsic productivity has a lower extinction risk than a population with low intrinsic productivity. This is because a population with a high intrinsic productivity is likely to return to high abundance if pushed to low abundance by environmental variation, whereas a population with a low intrinsic productivity is more likely to stay at lower abundance, making it more susceptible to extinction during the next period of poor environmental conditions. Although a convention of population biology, the term intrinsic productivity may be somewhat misleading in that it suggests the value is an inherent property of a species, when in fact it is a function of both the species’s biology and the environment (which can change).

A generic approach to identifying a viable productivity-abundance criterion is to estimate extinction risk using a population dynamics model and determine the threshold at which productivity and abundance parameters just yield an acceptable risk. The results of these sorts of analyses can be plotted in a viability curve, on which every point represents a productivity-abundance combination with identical extinction risk (Figure 4.1). Two key issues in developing

a specific method from this generic approach are defining the form of the population dynamics model used to estimate extinction risk and determining the method for estimating the model parameters. We explored a number of different functional forms for the population projection model and methods of estimating parameters (Appendices D–H). All the projection models we examined are variations of a spawner-recruit model. Spawner-recruit models use a relatively simple function to predict the average number of recruits produced by a given number of spawners. Recruits can be defined in terms of different life stages (e.g., JOM, pre-harvest, returning spawners). Figure 4.2 illustrates a number of different potential spawner-recruit relationships. The slope of the curve at the origin (near 0 spawners) is considered the intrinsic productivity of the population as defined above and is of critical importance in estimating extinction risk. A central distinction among the different approaches we explored to set criteria is the method used to estimate this productivity parameter. Our analysis suggests criteria not be based on a single method, but rather on a hierarchical approach to parameter estimation that is driven by the information content of the data. Where the data allow, an approach that involves fitting spawner recruit curves may be more appropriate; where the data are sparse, the PCC approach may be most appropriate. Both approaches are described below.

In using PVA models to define a viability curve (Figure 4.1), we attempt to identify threshold conditions that just produce an “acceptable extinction risk.” A statement of acceptable risk may be phrased as “an X% probability of declining to lower threshold of spawners in Y years.” The lower threshold could be either true extinction (i.e., 0 fish) or a quasi-extinction threshold (QET). The QET represents an abundance below which the population should not go because it would experience a greatly elevated extinction risk as the result of processes other than environmental stochasticity, or because uncertainty about population behavior is highly elevated. Two factors contributing to highly elevated extinction risk at very low abundance are demographic stochasticity and increased risk of permanently losing genetic variability. The X and Y values in the risk statement are largely policy decisions about what is legally and socially acceptable. Guidance from NOAA Fisheries suggests a 5% probability in 100 years is appropriate for defining a viable population threshold (*sensu* McElhany et al. 2000). A number of different time periods are discussed in the context of these criteria (Table 4.1).

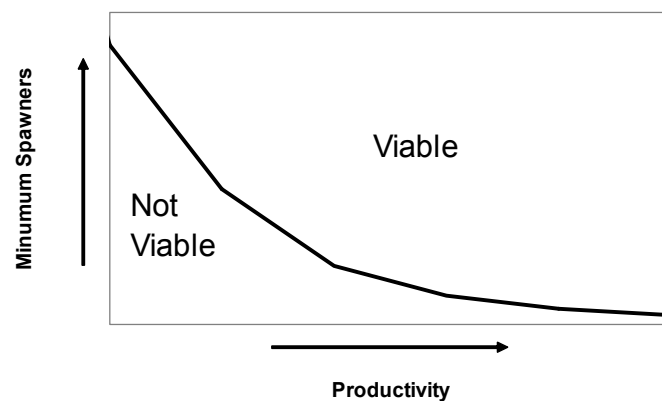


Figure 4.1 Conceptual graph of the relationship between productivity, population size, and extinction risk. The curve represents combinations of size and productivity that exactly have the acceptable extinction risk.

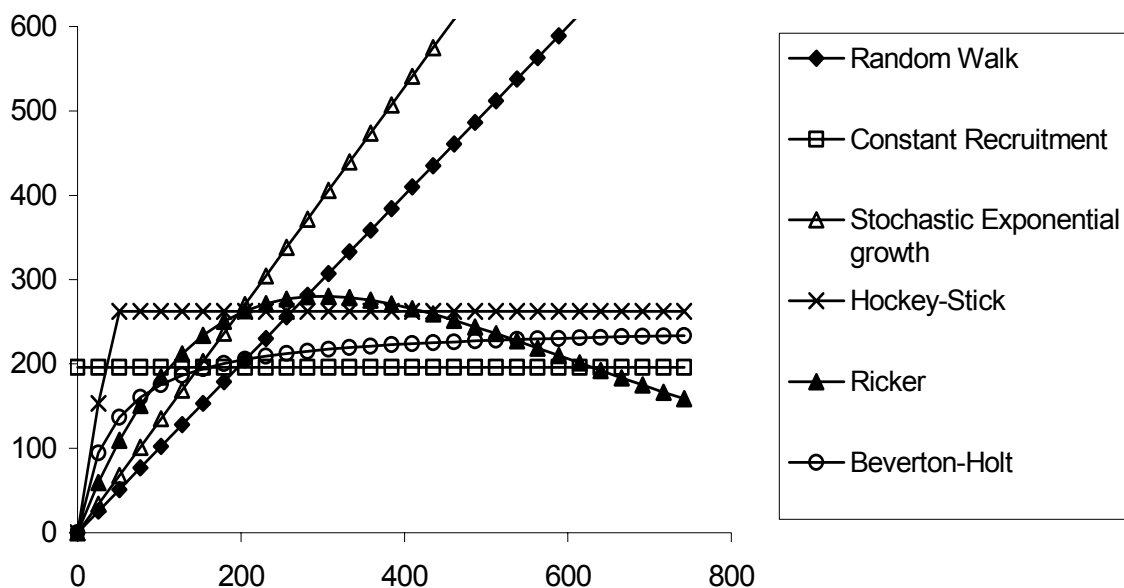


Figure 4.2 Different types of spawner-recruit curves.

Table 4.1 Definitions of time intervals used in discussion of population productivity and abundance criteria.

Time Interval or Period	Definition
Historical period	Period just prior to Euro-American impact on salmon populations and habitat.
Recent time series	Period encompassing recently collected time series of abundances. Most time series start after the 1960s.
Observation period	This is the period over which a population will be evaluated to determine if it is viable. For delisting decisions, this is likely encompasses the period from the present to some point several decades in the future.
Extinction risk time horizon	This is the period over which we expect the population to persist with some probability. For PVA modeling, we evaluated periods of 100 years into the future
Running sum length	In setting the population change criteria, the exact method used relies on a running sum of the abundance data. The length of the running sum used for this analysis was four years. The current size and target size for the population change criteria are reported as the four-year average, not a running sum.

Fitting Spawner Recruit Curves

In fisheries biology, a common approach to estimating the parameters of spawner-recruit models is to statistically “fit” spawner-recruit curves to abundance data. Spawner-recruit data can be visualized by plotting the number of recruits against the number of spawners. Table 4.2 shows sample spawner-recruit data and a number of different curves fit to that data. To use a spawner-recruit analysis to estimate extinction risk, it is necessary to determine which, if any, recruitment functions provide an adequate approximation of the data and to determine the degree of confidence in the parameter estimates. In Appendix G, we attempt to fit a number of different potential recruitment functions to recent spawner-recruit data from salmon populations in the WLC, where recruits are defined as either returning spawners or preharvest adults. A key conclusion of this analysis is that the examined abundance data provide very little statistical power to estimate spawner-recruit relationships. That is, the data are not very informative about either the form of the recruitment function or parameters such as intrinsic productivity. This is unfortunate, because an accurate description of the spawner-recruit relationship, particularly the relationship at low abundance, would greatly aid in assessing population extinction risk.

The poor fit of the recruitment functions is likely due to a combination of factors including measurement error, environmental stochasticity (especially in the ocean), the lack of contrast in spawning escapement due to the constant infusion of hatchery spawners, and uncertainty about the reproductive success of hatchery spawners. One response to the problem of a poor fitting spawner-recruit curve is to try to improve the fit by collecting better data. Part of the reason that fitting spawner-recruit curves may be so uninformative in many populations is the high levels of measurement error in the abundance or age structure estimates. If the data had fewer errors, the spawner-recruit parameters could be estimated with better accuracy and precision. However, lower measurement error may not solve all the problems of fitting recruitment functions. Accurate parameter estimation requires a relatively large number of data points from a stationary time series, with adequate “contrast” in the spawner abundance. These issues are discussed in more detail in Appendix G.

Given that it may not be possible to precisely estimate productivity from fitting spawner-recruit curves with adult data for many populations, we considered several alternatives.

- It has been suggested that an average recruit per spawners value calculated using the low spawner abundance data points could provide important information about the resilience of a population (Chilcote⁴). This method does not fit a spawner-recruit curve and does not estimate “true” intrinsic productivity, but could provide a precautionary estimate of its value. The statistical properties of this approach, and the conditions under which it could be deployed, have not been rigorously explored, but the approach holds promise for development into a viability metric.
- By partitioning the projection model into multiple life stages, extinction risk may potentially be better assessed. Below and in Appendix G, a two-life-stage model is explored that partitions the life cycle into freshwater and marine life stages. Some variability that contributes to the poor fit of adult spawner-recruit curves can be accounted for in a multi-life-stage model, therefore parameters may be better estimated. Evaluating extinction risk with this type model requires accurate

⁴ Mark Chilcote, Oregon Department of Fish and Wildlife, Portland Oregon, personal communication, February 2003.

abundance estimates at multiple life stages (e.g., both adult spawners and juvenile outmigrants).

- Observed population growth rate can provide a precautionary estimate of the productivity of a population. This concept was developed into the PCC described below and in Appendices D–F. Although the method does not estimate the true spawner-recruitment relationship, it can be applied to any time series of abundance, and the statistical properties of the approach are reasonably well understood.

Two-Life-Stage Projection Model

Extinction risk is largely determined by the productivity or resiliency of the stock. Analysis of Columbia River chinook salmon marine survival patterns indicates that marine survivals do not randomly vary but follow a pattern at the decadal scale; that is, decades of high and low marine survival. This indicates that extinction risk is not likely to occur at random, but during periods of low marine survival. Therefore, extinction modeling using recruitment functions should be developed with marine survival, an index of marine survival, and/or another measure of ocean productivity. Appendix H explores an approach to setting viability criteria in which density dependence is assumed to occur in the freshwater life stage, and marine survival is considered a density-independent factor driven by the environment. Criteria developed using this approach would likely demonstrate that, based on the model, a population has sufficient freshwater productivity and capacity to persist in the face of hypothesized future marine survival patterns. With this approach, there is no single freshwater productivity and capacity target, as multiple combinations of productivity and capacity could produce identical extinction risks. Instead of a single *a priori* target, the approach could potentially be used retrospectively to evaluate whether a population has improved enough to have an acceptably high probability of persistence.

In Appendix H, the approach is applied to evaluate the current status of the Wind River steelhead population. Model parameters were fit using the spawner and smolt data, and a forward project of abundance was modeled under a number of hypothesized future ocean survival patterns. The Wind River data have not yet been analyzed using a formal model selection procedure like that described in Appendix I. The current Wind River data set consists of only seven data points, and it would not meet the standards of statistical rigor required for extinction analysis. However, as more data are collected, this approach could be used.

Population Change Criteria

Overview

The PCC approach is a novel method of developing viability criteria. With this approach, productivity is estimated from the observed growth rate of the population, not from fitting spawner-recruit curves. If a population grows at a given rate, it is assumed in the PCC approach that its average productivity is at least as high as the growth rate estimate. Because of the potential influence of density dependence, the population's intrinsic productivity may actually be higher than the observed growth rate, so the approach is precautionary in applying the criteria

and is unlikely to result in prematurely concluding that a population is viable. This approach addresses the question, “Given the current population size, what growth rate does the population need to exhibit over a given number of years to just achieve an acceptably low extinction risk?” The approach is a performance metric for assessing viability. Details on the approach are provided in Appendix D and a computer program to calculate the criteria is available on the Web at http://research.nwfsc.noaa.gov/cbd/trt/trt_wlc/viability_report.htm.

The forward projection model used for the PCC approach is a hockey-stick recruitment model. The PCC approach involves identifying the average growth rate for the population over Z years that just produces an acceptable extinction risk. As illustrated in Figure 4.1, the extinction risk of a population is a function of both the productivity and abundance of the population. As a consequence, the target growth rate (productivity) is a function of the population’s initial size. To calculate the extinction risk associated with a given combination of current abundance and growth rate, we must have estimates of all the parameters needed as input to the extinction risk model: environmental variance, QET, time horizon, initial size, productivity, and capacity. The environmental variance for a population is estimated from recent time series; the QET is set at 50 spawners, based on demographic and genetic concerns; and the time horizon was evaluated at 100 and 200 years. The estimate of productivity is based on the estimate of population growth rate. By knowing a population’s current size and how big it gets in a given amount of time, its target size, we can estimate its growth rate. The initial size parameter in the population extinction model is the target size of the PCC. The model is parameterized so that if the population achieves the growth rate target in the specified time, it would not need to continue growing but would still be considered viable if it stabilized at the final abundance. Calculating the target size requires estimating the environmental variance and the growth rate of populations. These parameters are estimates; there is uncertainty about their true value. This uncertainty is incorporated into the calculation of extinction risk by using the parameters’ probability distributions, not the point estimates. In this way, the target sizes and associated growth rate explicitly include parameter uncertainty. The criteria estimated with the PCC approach can be expressed either as target abundances or as growth rate. For ease of communication and because it better reflects the key parameter of the analysis, we present results in terms of observed growth rate.

The PCC is a population performance test rather than a statement of how many fish are needed for viability, and is in many ways different from other approaches the TRT considered. Understandably, a number of concerns were raised about the PCC approach; they are addressed in Appendix F.

Complications in Estimating Productivity Addressed with PCC

In the PCC approach, productivity is estimated as the change from the current population size to target size in a given amount of time (i.e., growth rate). This is a relatively straightforward calculation for a population of natural spawners. However, if hatchery-origin spawners are present in the population, the approach must be modified to estimate the natural productivity of the system. For a given acceptable level of risk, the target sizes are often substantially higher if hatchery spawners are part of the system than if they are not.

In the base calculations, we assume that the productivity, estimated as a change from the population’s current size to target size, is typical of the long-term average productivity. However, salmon respond to “regime shifts” in productivity, in which several decades of higher-

than-average marine survival may be followed by several decades of lower-than-average marine survival. To incorporate these regime shifts into the targets, the population productivity estimate is modified by a marine survival factor. This factor is a function of the difference between marine survival over the observation period and the long-term average. If marine survival over the observation period is higher than the long-term average, the target size needs to be higher than if the marine survival over the observation period matches the long-term mean. Because of uncertainties about marine survival patterns, this modification is applied asymmetrically: target sizes are raised if marine survival over the observation period is higher than average, but they are not lowered if the marine survival over the observation period is below the average. The marine survival modifications can only be applied after the observation period has passed, which makes it difficult to fix the target size at the outset.

Example Output of Population Change Criteria

An example output of the PCC approach is shown in Table 4.3. It is important to note that the growth rate targets are a function of the length of the observation period. The example in Table 4.3 uses an observation period of 20 years. Given the need to estimate population parameters with confidence, and the decadal scale shifts in marine survival described above, 20 years of data may be required before robust conclusions about viability can be made. A computer program for calculating population change criteria based on user-provided input is available on the Web at <http://www.nwfsc.noaa.gov/cr/programtest/salmonmodels.htm>.

In Tables 4.2 and 4.3, the extinction probabilities are the probability of declining to a four-year annual average of 50 spawners; they are calculated using population prediction intervals with 20 degrees of freedom for the variance estimate. The point estimate of the variance

Table 4.2 Growth and abundance viability criteria expressed as growth rate. The percent risk is the probability of declining to a four-year annual average of 50 spawners within 100 years.

Starting Population Size ^a	Average Growth Rate Observed Over 20 Years ^b			
	60% Risk (Persistence Category 1)	25% Risk (Persistence Category 2)	5% Risk (Persistence Category 3 (VSP))	1% Risk (Persistence Category 4)
<150	200 spawners	400 spawners	800 spawners	1,400 spawners
150–500	2%	6%	11%	15%
500–1,000	–1%	4%	9%	13%
1,000–1,500	–2%	2%	8%	12%
1,500–2,000	–3%	2%	7%	12%
2,000–3,000	–3%	2%	7%	11%
3,000–4,000	–3%	1%	7%	11%
4,000–6,000	–4%	1%	6%	11%
6,000–8,000	–4%	0%	6%	11%

^a The starting population size is estimated at the beginning of the period being evaluated for viability.

^b The productivity viability metric is the average annual productivity that would need to be observed over a 20-year observation period. Modification of the productivity would be required if hatchery fish are present or if the observation period occurred during a period of higher than average marine survival.

Table 4.3 PCC productivity criteria for conditions where hatchery fish are present or marine survival over the observation period differs from the long-term average.^a

Starting Population Size ^b	Average Growth Rate of Natural-Origin Spawners Observed over 20 Years ^c			
	Effective Fraction of Hatchery-Origin Spawners ^d			Assuming Ocean Survival Is Twice Long-Term Average ^e
	5% Hatchery	10% Hatchery	30% Hatchery	
<150	1,400 spawners	2,900 spawners	>3,000 spawners	1,300 spawners
150–500	15%	20%	>21%	14%
500–1000	13%	19%	>21%	13%
1000–1500	13%	18%	>21%	12%
1500–2000	12%	17%	>21%	11%
2000–3000	12%	17%	>21%	11%
3000–4000	11%	16%	>21%	11%
4000–6000	11%	16%	>21%	10%
6000–8000	10%	15%	>21%	10%

^a The extinction risk associated with these criteria are 5% in 100 years, the same as in column four in Table 4.2.

^b The starting population size is estimated at the beginning of the period being evaluated for viability.

^c The productivity viability metric is the average annual productivity that would need to be observed over a 20 year observation period.

^d If hatchery fish are present and effectively spawning, the observed growth rate needs to be higher because of hatchery masking effects.

^e If the marine survival over the observation period is higher than the long-term average marine survival, the observed growth rate needs to be higher to provide an equivalent long-term extinction risk.

used to generate these targets is 0.05. The current abundance values for WLC populations are shown in Appendix D. The growth rates in Table 4.2 assume that 0 hatchery-origin spawners are present in any of the populations in the next 20 years. (If hatchery-origin fish are expected, see Table 4.3 for examples). The targets also assume that the average of the marine survival index in the next 20 years is equal to long-term average marine survival.

Conclusions Regarding PVA Modeling Approaches

We recommend a data-driven approach to PVA-based viability criteria. Where data are sufficient, fitting spawner-recruit curves can provide a good retrospective analysis of population viability. Since a number of potential spawner-recruit curves could potentially be viable, we are not recommending a single curve as a viability target. In many cases, the collection of data on multiple life stages could provide a better estimate of population viability than a spawner-recruit curve that concentrates only on adults. Again, evaluation of viability would be retrospective, and we are not recommending a single set of parameters for a multi-life-stage model as criteria. The approach of estimating productivity and abundance target combinations based on estimates of average recruits per spawner using low-abundance data points has promise, but is not yet fully developed. The PCC have the least data requirements and can be estimated in advance to provide target criteria for most all populations. However, the PCC may be overly precautionary in some cases, and an effort should be made to apply one of the other methods if data are adequate. Since

the PCC can be applied to all populations based on current information, and viability with the other approaches will likely be only evaluated retrospectively, we recommend that the PCC serve as default criteria.

Historical Abundance

Historical Abundance Estimates Using Broad-Scale Habitat Analysis

Historical abundance can potentially be estimated from a study of historical habitat quantity and quality. Such analyses are likely to produce results with a high level of uncertainty because they require estimating both historical habitat conditions and associating fish abundance with habitat condition. Nevertheless, historical habitat analysis can inform viability criteria by suggesting some upper bounds on target abundances. The HPVA discussed below attempts this type of calculation using a relatively large number of habitat variables that require estimating a relatively large number of quantitative relationships between habitat attributes and population response. The broad-scale analyses conducted by Steel and Sheer (Appendix I) estimates fish densities implied by a range of population viability criteria for both currently and historically available habitats. These analyses are based on relatively few habitat attributes, which can be estimated from available data.

The approach to broad-scale analysis undertaken taken by Steel and Sheer partitions the WLC domain into different habitat types based on remotely sensed data and on digitally available and spatially referenced field data. These data allow habitat partitioning based on features such as accessibility, stream gradient, stream width, etc. It would be theoretically possible to estimate historical abundance of a population by multiplying an estimate of the fish density associated with each habitat type by the quantity of that type available, then summing all the different habitat types. However, such an approach would require good estimates of the species densities associated with each habitat type, and the analysis could become very complex as issues of density dependence at different life stages are introduced.

Rather than estimate historical abundances per se, we have taken the approach of estimating population targets via demographic modeling, then evaluating whether the targets would be associated with reasonable historical fish densities. In attempting to identify reasonable fish densities, we can divide fish density into three categories: (1) clearly historically achievable, (2) clearly historically unachievable, and (3) historical achievability unknown. It should be possible to identify these regions without having to specify all the parameters needed to generate an estimate of historical abundance. The regions could be identified based on observations of currently “healthy” populations. If the population targets developed by demographic modeling fall into category 1, we will accept the demographically derived target as the criteria. If the target falls into category 2, we may need to look more carefully at the demographic modeling assumptions. If the target falls into category 3, further analysis will be required to evaluate the historical abundance of the population.

Thus far, habitats throughout the WLC domain have been categorized into types based on a number of features. Tables were developed that estimate the densities associated with some example population targets, based on channel gradient and on currently and historically accessible stream lengths. We have not yet evaluated whether these densities are reasonable. Furthermore, the process of categorizing the habitat is undergoing refinement. As the approach

develops, it may ultimately be possible to provide the “credible estimates of historical abundance” described in the viability criteria for situations in which the historical abundance may have been lower than the demographically established target.

Historical Abundance Estimates Using HPVA

Habitat population viability analysis (HPVA) is a specific application of the ecosystem diagnosis and treatment (EDT) method (Lichatowich et al. 1995, Mobrand et al. 1997) currently used in recovery planning. It uses up to 45 habitat and landscape attributes to predict abundance, productivity, spatial distribution, and diversity of a population under particular habitat conditions. The basic method for running an HPVA of a watershed for a particular species is as follows:⁵

1. Stream reaches are defined by delineating the geographic scope, describing environmentally homogeneous reaches, and coding the basin hydrography, indicating the direction of water flow and the spatial relationship of tributaries in such a way that it can be understood by a computer program.
2. Information on the species' life-history parameters is incorporated into the model. These include juvenile age at migration, juvenile migration pattern (spring or summer migrant, etc.) adult age at return, run-timing, ocean distribution, harvest rate and location, fecundity, and number of females by age.
3. As many as 45 habitat attributes known to affect salmonid performance are included in the database. Attributes include percent habitat types, stream substrate, channelization level, riparian condition, water quality and quantity, percent fine sediment, toxic substances, exotic species present, food, large woody debris, and many others. If measured data are not available, either professional opinion is used or the attribute is ignored altogether. In current analyses used by Washington State and tribal co-managers, four tiers of attributes are used: current conditions, PFC conditions, PFC+ conditions, and historical conditions. PFC conditions are attribute ratings based on the properly functioning condition values in the Matrix of Pathways and Indicators (NMFS 1996). PFC guidance for estuarine and marine habitats does not yet exist. The condition of these habitats was set at current for one analysis of otherwise PFC conditions (called HPVA_{PFC}), and at fully functional (historical) for another (called HPVA_{PFC+}).
4. The model is now run. Trajectories are used to estimate fish survival from egg incubation all the way through adult spawning. Briefly, a trajectory starts out in identified spawning reaches, then is moved through time and space by life stage as determined by the species' biology. Hundreds of these trajectories are sent from a basin to determine its survival landscape. Randomness is included in the trajectories so that the full range of environmental conditions present in the stream is encountered. The model calculates and tracks the productivity and capacity of each trajectory through each stream reach. Trajectories that have productivity less than 1.0 are nonviable, thus are considered not to be used by the fish population being

⁵ Modified from a description by Bruce Watson of Mobrand Biometrics, Inc.

modeled. The productivity and capacity values of all trajectories are combined to determine the productivity and capacity values for the population, assuming a Beverton-Holt relationship. The difference in the number of successful trajectories under various conditions (e.g., historical versus current) is used to calculate a diversity index for the population. A run is done for each of the four tiers of habitat attributes, so productivity, capacity, and diversity are estimated for current, PFC, PFC+, and historical conditions.

Output from analyses run on Lower Columbia River populations are shown in Appendix H. HPVA data are only available for populations in Washington, therefore complete coverage is not available for any of the WLC ESUs. Mobrand Biometrics, Inc. cautions against using the EDT model for predictive purposes: they state that its intended application is hypothesis generation. Therefore, although no confidence metrics are provided, values provided in the table should be considered highly uncertain. However, they do provide some hypotheses about historical abundance against which to compare the PVA model estimated targets.

Risk Characterization

Using PVA models provides the greatest opportunity to quantitatively relate any of the criteria to the 0–4 population persistence categories. For example, Table 4.2 shows how PCC could be related to persistence categories. If data are available, the other PVA models discussed above could also be used to calculate persistence probabilities. In addition to the PVA model approaches discussed above, other data, such as presence-absence information, could inform an evaluation of the risk status of a population. However, the other types of data generally have greater uncertainty regarding their relationship to viability, and we are not recommending their use as primary criteria. Any risk characterization that utilized metrics not clearly related to viability would likely result in the population being considered in a lower category because of increased uncertainty. As described in the criteria bullets, the estimate of historical abundance could be important in assigning the persistence category.

5. JUVENILE OUTMIGRANT GROWTH-RATE CRITERIA

JUVENILE OUTMIGRANT PRODUCTION CRITERIA GUIDELINES
1. The abundance of naturally produced juvenile outmigrants should be stable or increasing as measured by observing a median annual growth rate or trend with an acceptable level of confidence.

Definitions

Juvenile outmigrants (JOMs) are fish that are leaving a watershed. These are not necessarily smolts; the exact life stage depends on species and life history.

JOM abundance is based on cohort age for populations with multiple age classes in the outmigrating population. Abundance may be an estimate of total abundance or a standardized index of abundance.

JOM growth rate (λ_{JOM}) is the annual change in cohort-to-cohort abundance (whole population or index), estimated with a four-year running sum or a fitted trend (slope).

Overview

The TRT strongly supports the inclusion of JOM growth-rate criteria for assessing the viability of salmon populations. The criteria will contribute important information about the status of a population that cannot be obtained by exclusively monitoring adults. JOM monitoring has multiple benefits and should be a priority in all ESUs, particularly for populations in which freshwater habitat improvements are a major management goal. However, it is not clear whether all populations in an ESU need to be extensively monitored or meet the JOM growth-rate criteria. JOM monitoring should, at a minimum, be part of recovery planning for all populations designated to attain VSP status in an ESU.

Viability criteria based purely on abundance and spawner trends are problematic. Spawner numbers fluctuate for a variety of reasons, including harvest management, the influence of hatchery spawners, and long-period ocean productivity cycles that affect smolt-to-spawner survivorship. Populations may increase over a relatively short period due to good ocean conditions, even while freshwater productivity declines. This factor is particularly important for short-term recovery assessment, because ocean conditions are currently favorable for many populations and the number of returning spawners may increase rapidly. Spawner counts also give a poor indication of how well management actions are improving survival at various life stages. Efforts to improve freshwater habitat and survival will be difficult to evaluate without a benchmark at some other point in the life history. JOM production can serve as this benchmark and must meet minimum standards *in addition to* population growth and abundance criteria set for adults.

Several studies have shown a relationship between salmonid abundance or ocean survival rates and periodic shifts in the physical and biological characteristics of the North Pacific, such as the Pacific Decadal Oscillation (PDO) (Hare et al. 1999; McFarlane et al. 2000; Hobday and Boehlert 2001). Patterns in ocean survival can have a profound effect on the number of returning spawners with periods of rapid population increase that can mask changes in freshwater capacity and productivity (Lawson 1993; Bisbal and McConnaha 1998; Tschaplinski 2000). Figure 5.1 illustrates how spawners and JOMs might vary in a population experiencing cycles in marine survival. Measures of JOMs and adult spawners may provide different types of information that are critical for assessing the long-term viability of populations. While JOM abundance should track adult abundance when the population is below carrying capacity, changes in freshwater habitat quantity or quality are better assessed by JOM productivity.

JOM monitoring is required for estimates of marine survival rates. Because the productivity and abundance criteria (Chapter 4) require a correction for ocean survival rate, a minimum number of populations must be monitored to ensure that the correction factor is

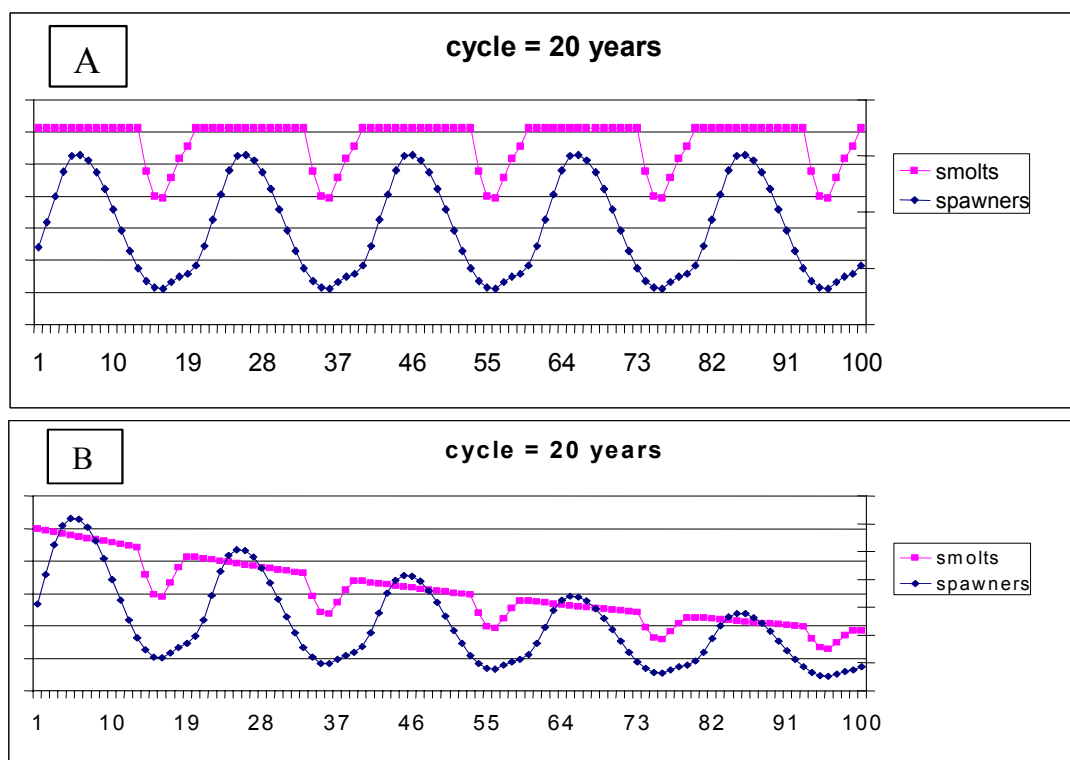


Figure 5.1 Hypothetical changes in spawner and smolt abundance in a 20-year cycle (sine-wave) of ocean survival. To make this example easy to interpret, no variability was added to the simulation. A hockey-stick, smolts-per-spawner function was applied. A. Freshwater capacity is constant through time. B. Freshwater capacity is declining at 1% per year. Note that the spawner growth rate remains positive for a much longer interval than the JOM growth rate; the difference depends on how quickly the recovering population reaches carrying capacity.

relevant to a given population. Currently, marine survival rates are primarily calculated with hatchery fish returns; this is insufficient, unless we are certain that marine survival rates for hatchery and wild fish are comparable. Likewise, because marine survival encompasses survival in the mainstem Columbia and estuary, we should expect that the rates would vary by stratum ecoregion and life-history type within an ESU. JOM monitoring in multiple populations is thus a critical part of population assessment and an important indicator of ESU viability.

Approaches Considered

The production of JOMs is an indicator of freshwater habitat productivity and capacity (Nickelson and Lawson 1998; Bradford et al. 2000; Sharma and Hilborn 2001). Although environmental factors and measurement error make JOM abundance highly variable from year to year, it is critical to assess the JOM growth rate and confirm that productivity is not decreasing in recovered populations. A decreasing trend in JOM abundance over a long period suggests deteriorating habitat quantity or quality. An exception to this would be a decrease in JOMs due to declining spawner abundance; in that case, a population would not be considered viable under the spawner growth and abundance criteria. Additionally, JOM abundance could decline if there was a decrease in the average age (and size) of spawning females. Under any of these three scenarios, a long-term decline in JOM abundance indicates a nonviable population. In computer simulations that include cycles of ocean survival rates, the JOM growth rate is consistently the best metric for correctly assessing changes in freshwater productivity and capacity when compared to other potential metrics, such as JOM/spawner or absolute JOM abundance (two-stage model, Appendix H). However, as with spawner abundance and growth, this metric of population health and recovery is only an accurate indicator when calculated over a relatively long period. This is because growth rate estimates and spawner abundance are both variable and are correlated.

Based on our current understanding of ocean productivity cycles, a minimum of 20 years of JOM abundance estimates are considered necessary to obtain an accurate assessment of population viability. Longer time series of λ_{JOM} should also increase confidence that a decline in productivity, freshwater capacity, or juvenile survival is not occurring.

Several other criteria for juvenile production were considered, including a minimum JOM/spawner, a minimum JOM population size (JOM_{min}), and a requirement for $\lambda_{\text{JOM}} \geq \lambda_{\text{spawner}}$. JOM/spawner may provide valuable information during early stages of recovery, when rearing habitat is below full seeding capacity. If freshwater habitat capacity and productivity are not in decline, we would expect no change in JOM abundance if spawning escapement is high enough to seed habitat to capacity, or an increase if spawning escapements have been lower than seeding levels. However, JOM/spawner may decrease dramatically as the juvenile population reaches freshwater carrying capacity; this may or may not indicate a problem in the freshwater habitat. If the JOM abundance is declining, then it is likely at least one of the following is occurring: (1) freshwater habitat quality or quantity is declining, (2) seeding levels (number of adult spawners) are declining below the level needed to produce the maximum JOM abundance, and/or (3) a Ricker relationship holds and high seeding levels are causing a decline in JOM abundance. If JOM are declining *and* JOM/spawner is declining, the most likely cause is a decline in freshwater habitat quality and quantity. However, at this time, the TRT feels that λ_{JOM} is the best

metric because of uncertainty about the effects of ocean cycles on adult survival rates and assumptions regarding the current and future carrying capacity of our watersheds. While JOM/spawner may be a valuable metric during the early stages of population recovery and restoration, density-dependent changes in survival can drastically alter this relationship over time. An accurate estimate of JOM/spawner may also be difficult to obtain and highly variable due to measurement error and monitoring locations.

Additional simulation exercises and population-specific productivity data may make it possible to develop more specific JOM criteria in the future. Many managers are predicting that both management actions and improving ocean conditions will lead to increases in adults and juveniles over the next few years. By monitoring JOM production, we will be better able to separate the causes of population change and evaluate habitat restoration activities. More generally, delisting criteria and population assessments for JOMs are the first steps toward monitoring and evaluation of the entire salmon life cycle. JOM counts are already conducted at a number of sites. While JOM surveys are difficult and potentially costly, they are feasible for many populations, and may ultimately save resources by improving our ability to assess both salmon populations and management actions.

Strategies Selected

The abundance of juvenile outmigrants in viable populations should be stable or increasing with an acceptable level of confidence. For populations with life histories that include multiple age classes in the annual JOM count, abundance should be determined by cohort year rather than the sum of JOM from multiple cohorts. The JOM growth rate can be estimated as:

$$\hat{\lambda}_{JOM} = e^{\mu_{JOM}},$$

$$\mu_{JOM} = \text{mean} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right)$$

where N_t is a 4-year running average of JOM abundance, which may be an estimate of total JOM produced in a cohort year or an index of abundance that has been standardized for all populations within the ESU. The JOM growth rate should be determined by a running sum of 4 years to reduce variability caused by cohort strength. For example, if the assessment period is 20 years, only 16 estimates of N_t are used to determine the growth rate λ_{JOM} . In viable populations, λ_{JOM} should be ≥ 1.00 .

Alternatively, a trend in JOM abundance can be determined by regression analysis or a time-series analysis that incorporates autocorrelation. Each method will result in different confidence intervals and should be standardized within the ESU. A trend analysis may result in an acceptable level of confidence for $\lambda_{JOM} \geq 1$ in less than 20 years. However, we strongly advise long time intervals for assessment, due to the potential for decadal changes in marine survival rates.

If sufficient monitoring is in place to estimate spawner and JOM abundance by cohort year, a marine survivorship rate can be calculated simply as:

$$\text{marine survivorship} = \frac{\text{Spawners}}{\text{JOM}}$$

This marine survivorship estimate requires cohort-specific estimates of actual abundance, rather than indices of abundance. The productivity and abundance criteria require an estimate of marine survivorship at some level for an ESU, preferably for each stratum and optimally for each population, as populations vary in the amount of time spent in the ocean and estuary.

Critical Uncertainties

JOM abundance and growth rates may indicate freshwater habitat conditions, but the mechanisms of this relationship are still poorly understood. JOMs respond to habitat quality or quantity and environmental variability in unpredictable ways. The relationship between habitat quality and juvenile survival or other population-level metrics has not been assessed for most populations. Finally, most assessments of population response to density and habitat quality have been done in recent years, when stocks were declining or severely depleted. Improved ocean survival rates may drastically alter the number of returning spawners; management efforts that improve freshwater habitat quality may require new analyses of density-dependent relationships and changes in JOM productivity.

Computer simulations were used to evaluate a number of different JOM delisting criteria, including JOM/spawner minima and JOM abundance criteria. The goal of these exercises was to determine how often a given model correctly advises delisting under a wide range of “real life” scenarios, such as cyclical ocean conditions and various forms of density dependence. These exercises include a range of stock-recruit functions and other assessments of uncertainty, but they need to be improved and updated with new information. The simulations should include measurement error and explore the optimum criteria for a range of life-history types. Measurement error may be high for JOM monitoring, except in the rare cases where dam passage allows complete counts (Bradford et al. 2000; Phillips et al. 2000). Managers will need to develop appropriate monitoring and extrapolation methods that can be standardized within a population and, ideally, for all populations within the ESU. This will be a difficult task, as salmon life histories can be extremely complex. For example, several major life-history trajectories have been identified for spring- and fall-run chinook salmon. Chinook salmon and steelhead JOMs monitored at the mouth of a Lower Columbia River tributary will be from multiple cohorts due to the multiple life-history pathways in these species. Monitoring will require subsampling of JOMs to verify age-class and relate JOM abundance to cohort strength. This may be difficult if both fall and spring chinook JOMs are present in a watershed.

Monitoring and Evaluation

Population monitoring is an essential part of recovery planning and assessment. The TRT recognizes the potential difficulties of monitoring JOM production in all watersheds. These recommendations may be modified for each ESU, but represent our best professional judgment for general guidelines.

1. JOM abundance should be assessed annually in at least one population per stratum.

Monitoring should be integrated with efforts to assess changes in population diversity, habitat quality, and within-population spatial distribution, as well as efforts to monitor effects of restoration activities. This monitoring need not be restricted to JOM trapping; alternative mark-recapture estimates or other indices of abundance can provide good trend estimates, provided that methods are standardized among years.

Currently, JOMs are not censused in most populations. This lack of information reduces our confidence in stratum and ESU risk characterization, and will be reflected in the integration of population attributes used for population, stratum, and ESU assessment (Appendix C). The TRT strongly recommends the establishment of JOM monitoring at some level in all watersheds.

2. Methods for JOM assessment must be standardized within populations.

Rigorous monitoring and assessment methods for JOMs need to be established. There are three sources of variability to consider:

1. interannual variability, primarily due to environmental factors,
2. intra-annual variability among sites within a population, and
3. variability in estimate of JOMs at a given site due to methodology (e.g., mark-recapture estimation).

Estimates of abundance can be determined from mark-recapture of JOMs caught in traps, direct counts at dams, and index abundance measures. Most estimates of JOM are made using a trap efficiency method (Dempson and Stansbury 1991; Thedinga et al. 1994). Methods should be standardized in all cases where data are to be pooled, such as indices derived from multiple tributaries in a watershed, and general methods should be standardized for each ESU to allow statistical comparisons of population status. New statistical methods of smolt trap mark-recapture may be useful for standardizing JOM abundance estimates (Bjorkstedt 2000). Finally, if hatchery fish are present in the monitoring area, they must be marked to provide an accurate assessment of wild fish recovery.

It is unlikely that we can achieve an accurate estimate of total JOM production for an entire population, except in the rare case of a population that must pass through a dam. The best estimates of abundance may be obtained in small tributaries, but they may not be representative of the population. Thus, a pooled index of abundance may be the most feasible option. One approach that may provide good coverage across a stratum, with an adequate level of monitoring for trend estimation, is a rotational monitoring scheme that would measure abundance in multiple populations but staggered over multiple years. It is difficult to anticipate the level of uncertainty and measurement error *a priori* for each population or stratum, as so few populations are currently monitored. It may be possible to select an optimal strategy in the future, using a power analysis for predicting the maximum level of error in abundance that will lead to a detectable trend over various time intervals and number of sites. Figure 5.2 hypothetically illustrates what such a power analysis might look like for two levels of λ_{JOM} .

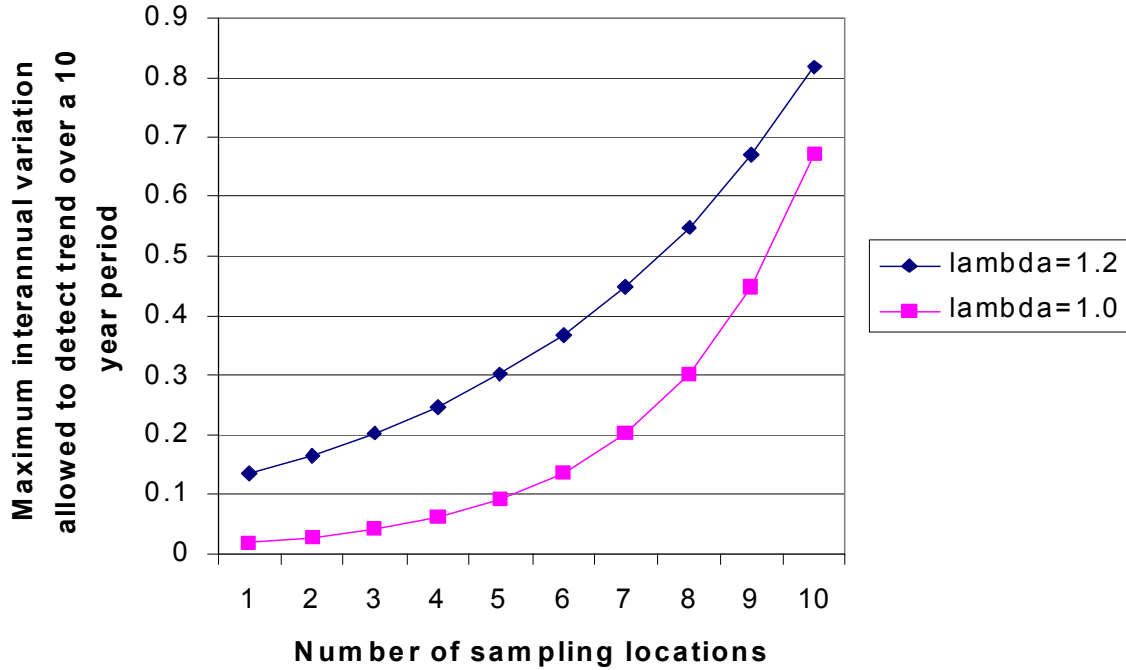


Figure 5.2 Hypothetical changes in the maximum intra-annual variability allowed for detection of λ_{JOM} , given an increasing number of samples. Detectability and confidence will depend on the number of years included in the analysis, as well.

3. An acceptable level of uncertainty in the growth rate estimate should be determined for each population.

Measurement error and year-to-year variability in JOM counts may lead to large confidence intervals for λ_{JOM} . Requirements for the appropriate confidence intervals or λ_{JOM} will depend on data quality and the methods used to determine JOM abundance in each population. Pooled counts from multiple tributaries in a watershed may reduce this variability. Most JOM abundance estimates for listed populations are likely to fall within plus or minus 25% because of the small sample sizes (fish abundance is low because they are listed) and low trap efficiencies (we tend to fish in the lower end of the basin in less flow to estimate the total population, and under these conditions trap avoidance is often high) (Schwartz and Dempson 1994). Within-year variability will be less critical with longer time series.

Figure 5.3 gives an example of three scenarios that might arise in an assessment of λ_{JOM} . For each example, the λ_{JOM} from a running sum and the λ_{JOM} estimated from a regression analysis are given. In Figure 5.3a, data variability within years is extremely high, leading to wide confidence intervals around each annual estimate. Figure 5.3b gives an example where within-year variability is moderate, but the number of years in the assessment is also low, reducing our confidence in λ_{JOM} . Figure 5.3c increases the number of years in the former example, and gives a good estimate of λ_{JOM} .

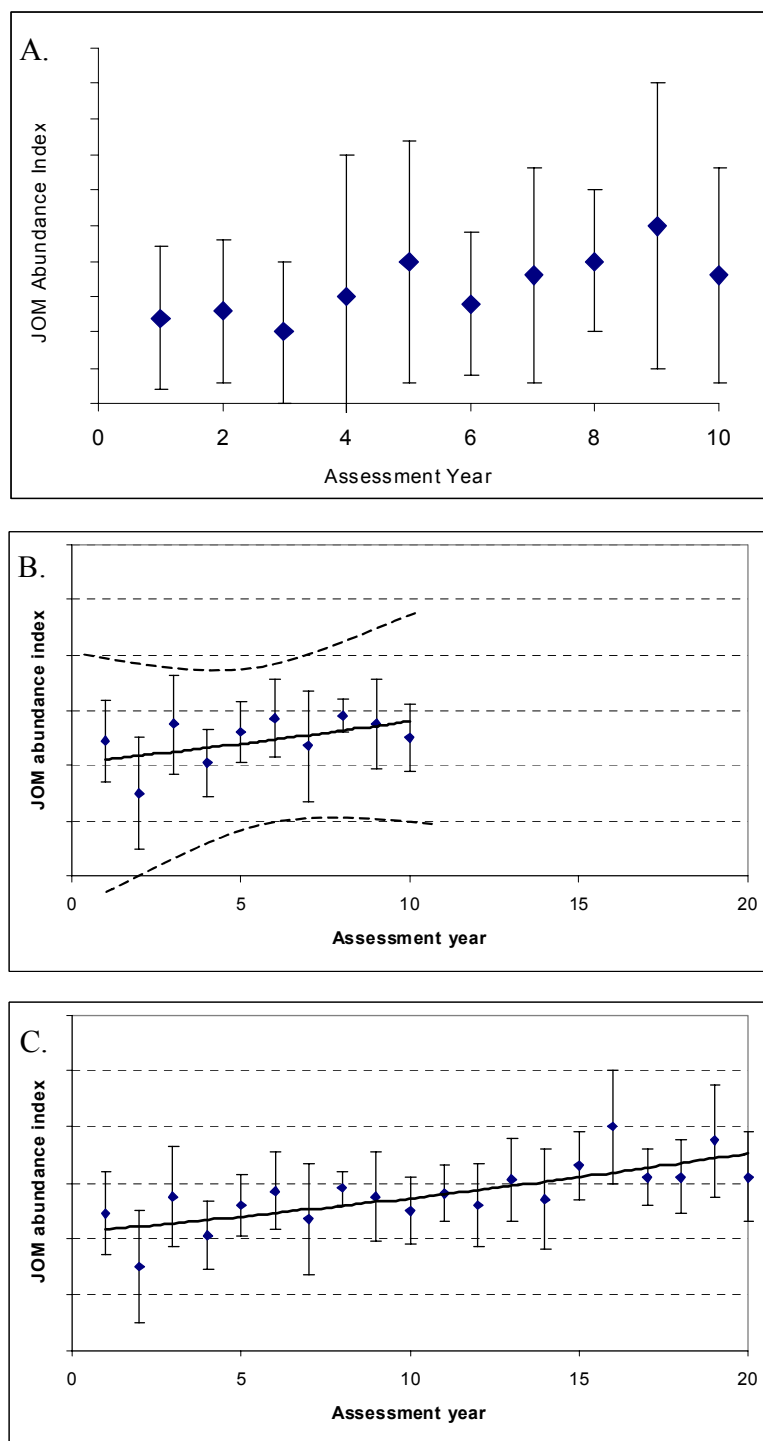


Figure 5.3 Three JOM abundance estimate scenarios, showing the importance of intra-annual variability for trend estimation. A. Visually, an increasing trend, but very high uncertainty in the estimate for each year reduces our confidence—this population could even be decreasing. B. A scenario with greater confidence in the abundance estimate each year, but too few years to be statistically confident in the trend. The dashed lines represent 95% confidence intervals for the slope given by the solid line. C. When additional years are added to scenario B, we are much more confident that the JOMs are increasing for this population.

Once a marine survivorship rate has been calculated for a population, it could be assumed to apply to other populations in that stratum. Thus, given the estimated spawner count by cohort year and a marine survival rate, the number of JOMs produced by a population could be back-calculated as follows:

$$\hat{JOM} = \frac{\text{spawners}}{\text{marine survivorship}}$$

This assumes that (1) the marine survivorship rate for one population in a stratum is the same as for another population in a stratum, and (2) the number of returning spawners in a cohort can be accurately estimated. Because the extrapolation of a marine survival rate to neighboring populations adds uncertainty to the JOM estimate, the JOM criteria score is discounted (see “Integrating with Other Criteria,” below). As more information on JOM trends and marine survivorship become available, a correlation analysis may help determine the validity of extrapolation. Also, we note that there will be a lag of as many as seven years required to estimate chinook marine survivorship, as the rate estimate requires that all potential spawners from a cohort have returned.

Risk Characterization

The relative value of JOM monitoring for evaluation of population status depends on data quality (accuracy) and how well the index or abundance estimate represents the entire population. We recommend that a scoring system for λ_{JOM} incorporate both the quality of the estimate and the predicted growth rate.

Table 5.1 gives an example of how population persistence category could be based on combinations of λ_{JOM} and statistical confidence in λ_{JOM} . The variance in JOM abundance estimates or indices between years due to environmental stochasticity is not related to data quality, but may make us less confident in a trend. The actual confidence levels used to determine population persistence category will depend on the acceptable level of risk.

Table 5.1 Relationship between λ_{JOM} and population persistence category.

Population Persistence Category	λ_{JOM} and confidence
0	Declining with high confidence in slope or extrapolated from other data sources
1	Stable, extrapolated from other data sources
2	Stable or increasing, low confidence in trend or extrapolated from other data sources
3	Stable or increasing, medium confidence in trend
4	Stable or increasing, <u>high</u> confidence in trend

Data quality for population attribute integration (Appendix C) may not be directly related to the confidence interval calculated for λ_{JOM} and should be assessed through professional judgment. Factors to consider include standardization of methods across a watershed, the proportion of the population represented in the index or indices used to estimate λ_{JOM} , and whether λ_{JOM} for the given population was assessed directly or extrapolated from other sources (for example, estimates from neighboring populations or from an adult spawner estimate with a marine survivorship calculated for a neighboring population).

If wild fish marine survivorship can be derived from JOM and returning spawner abundance, it can be used in the productivity and abundance criteria for multiple populations within a stratum (Section 4). Changes in JOM abundance can also serve as quantitative indicators of habitat quality (Section 7).

6. WITHIN-POPULATION DIVERSITY CRITERIA

WITHIN-POPULATION DIVERSITY CRITERIA	
1.	Sufficient life-history diversity must exist to sustain a population through short-term environmental perturbations and to provide for long-term evolutionary processes. The metrics and benchmarks for evaluating the diversity of a population should be evaluated over multiple generations and should include: <ul style="list-style-type: none">a. a substantial proportion of the diversity of a life-history trait(s) that existed historically,b. gene flow and genetic diversity should be similar to historical (natural) levels and origins,c. successful utilization of habitats throughout the range,d. resilience and adaptation to environmental fluctuations.

Overview

Genetic diversity, and the morphological and physiological traits that it determines, defines the life-history characteristics of a population and its ESU. In established populations, this life-history diversity reflects generations of adaptation to local environmental conditions and is fundamental to population sustainability. Similarly, biochemical measures of genetic diversity can reflect historical patterns of reproductive isolation and may be indicative of ancestral phylogenies; however, anthropogenic factors have done much to disrupt historical patterns of genetic diversity. Riddell (1993) states that genetic variation, within and between population levels, and productive habitats are the resource base of Pacific salmon, both for long-term sustainable production and continuing evolutionary processes. Population diversity is not a static attribute, but is representative of dynamic processes involving migration, mutation, genetic drift, and adaptive selection for an equally dynamic environment. The current depressed status of many salmon populations may be due, in part, to the loss of diversity or the disruption of the adaptive relationship between a population's life-history traits and its environment. Dramatic decreases in the abundance of a population, even for one or two generations, can result in the loss of genetic diversity. Local adaptation can be disrupted when nonnative conspecific fish are introduced into the breeding population, or when environmental changes occur at a substantially faster rate than the population can adapt to (especially when the new conditions are outside the normal range).

When considering the criteria for the level of within-population diversity needed to establish sustainable populations, we focus on diversity in major life-history traits and in life-history trajectories or strategies (those that most directly influence an individual's fitness). Consideration should also be given to the status of natural processes that create and maintain life-history diversity. Stearns (1976) defines life-history strategies as a set of co-adapted reproductive traits resulting from selection in a particular environment. Collectively, the expression of life-history traits in an animal allows it to successfully move temporally and

spatially through diverse ecological landscapes during the course of its life. For salmonids, major life-history trajectories include traits such as juvenile utilization of freshwater habitat, time of saltwater entry, age at maturation, and spawn timing and location (main stem, side channel, inlet, beach, etc.).

For example, salmon that spawn in short coastal systems generally need to emigrate before summer conditions (low flow and high water temperatures) render much of the river uninhabitable, or before barrier berms form at the mouths of the rivers. Some juveniles may move rapidly to the ocean, while others may remain in freshwater for weeks or months. Juvenile body size is also positively correlated with successful emigration to the ocean. Larger eggs produce larger juveniles, but females with large eggs have fewer total eggs. Older fish are larger and more fecund, but there are additional risks to remaining in the marine environment for an additional year. Thus, traits such as juvenile emigration timing, egg size, and age at maturation create a matrix of possible life-history options. The success of these options depends on the both the general basin ecology and the specific environmental conditions for that year. In addition, some species of salmonids (for example *Oncorhynchus mykiss* and *O. tshawytscha*) also display variability in anadromy, with both freshwater resident and anadromous phenotypes present in some populations. Whether resident fish are included in population status evaluations needs to be determined on a population by population basis. The U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) are currently developing criteria to evaluate the relationship between resident and anadromous forms within a population.

Sustainability depends on a population's ability to exploit available habitats, to adapt to environmental changes by maintaining multiple life-history trajectories, and to modify life-history trajectories in response to changing environmental conditions. Diversity provides a population with a repertoire of potential life-history trajectories. In this way, a population is able to effectively buffer itself against short-term environmental changes. Life-history diversity in association with the plasticity of life-history trait expression enhances (but doesn't necessarily ensure) a population's ability to remain productive through highly variable environmental conditions. Although there are probably an infinite number of possible life-history trajectories, previous selection and current local habitat conditions define a limited number of successful trajectories. Ecological differences among watersheds may result in differences in the suite of successful trajectories; therefore, there is no single set of diversity criteria that can adequately be applied across strata to all populations in this ESU. Historical indices of life-history diversity provide the most likely benchmark for a recovered population; however, the quality and quantity of historical information available is extremely limited. Examination of existing populations may be useful in defining diversity criteria, although the degradation of habitat conditions and population integrity may have altered or constrained the expression of life-history traits. It is also possible that different approaches to basin management and habitat restoration will result in different but equally sustainable complexes of life-history trajectories. Analysis of both historical and current life-history and habitat information will be necessary to identify those life-history strategies that are fundamental to a population's sustainability. The goal is to reestablish the natural (historical) relationships between habitat and life-history expression, rather than exactly recreate historical diversity patterns.

Developing specific goals for life-history diversity is daunting. It is not realistic to prescribe the relative contribution of any one life-history trajectory; therefore, a more general approach is warranted. A sustainable population needs to exhibit life-history traits (trajectories)

that are adapted to local conditions. Diversity criteria can focus on one or more aspects of the suite of life-history trajectories: the presence of specific life-history trajectories, the relative frequency of these trajectories during any given year, and a population's responsiveness (as measured by changes in the relative frequency of different life-history trajectories) to annual changes in environmental conditions. Regardless of the diversity metric(s) selected, the underlying issue of what level of diversity is sufficient to ensure population viability must be addressed. Although there are some general benchmarks that can be applied (e.g., historical levels), establishing specific recovery levels of diversity for a population will likely require work at the local recovery planning scale.

Approaches Considered

The TRT concluded that monitoring life-history strategies, or trajectories, provides the most useful metric for measuring population diversity. Several other metrics were considered. Biochemical metrics (allozyme or DNA analyses) of genetic diversity are useful for monitoring genetic changes in populations. Declines in genetic diversity for a population can be indicative of increased inbreeding. Monitoring genetic changes can be informative in detecting high rates of gene flow (straying) between populations. Additionally, biochemical measures of diversity between populations are useful in understanding local adaptation patterns. Loss of genetic diversity across an ESU is generally thought to have serious consequences on ESU viability. While biochemical measures of diversity have not been directly linked to specific aspects of a population's adaptability or fitness, changes in diversity can be indicative of changes in effective population size or gene flow between populations. Population census information can also be useful. Changes in the effective population-size thresholds provide another method for the conservation of genetic diversity. WDFW (1997), in its Wild Salmonid Policy, suggests that an effective population size (N_e) of 500 individuals is sufficient to prevent the loss of genetic variability, through inbreeding or population bottlenecks, over a long period. In general, the number of spawning individuals or breeders (N_b) is much larger than N_e . This is because not all adults observed on the spawning ground contribute to subsequent generations. McElhany et al. (2000) discuss the relationship between N_e and N_b and cite several estimates of minimum population (breeder) abundances (417 to 4,170 per year) for the maintenance of sufficient genetic diversity to ensure long-term persistence. Population size provides a relatively simple criterion for ensuring against the loss of genetic variation under specific conditions. Spatial and ecological heterogeneity considerations might result in a larger minimum population size or some degree of geographic structuring of the population. Simply maintaining genetic diversity will not ensure the expression of important life-history traits, nor will expressing life-history traits for which the corresponding habitat no longer exists ensure the population's sustainability.

The criteria for within-population diversity should also provide that the expression of major life-history trajectories within recovered populations will resemble, but not necessarily duplicate, those that occurred historically. The NMFS "Definition of Species" paper (Waples 1991) defines a population as distinct, for ESA purposes, if it (1) is substantially reproductively isolated from conspecific populations and (2) represents an important component of the evolutionary legacy of the species. This definition is important to within-population diversity. Despite substantial reproductive isolation, a population may be part of a larger metapopulation

(Hanski and Gilpin 1997). The diversity expressed within a population may depend in part on how it functions in a metapopulation. The evolutionary legacy of species is the genetic variability that is a product of past evolutionary events and represents the reservoir upon which future evolutionary potential depends. Restoring historical life-history trajectories or restoring the natural processes that affected populations, reestablishes the evolutionary process of populations within an ESU.

Given the plasticity of life-history traits, it may not be feasible to prescribe criteria for the expression of life-history variants. Some life-history trajectories may only be detectable under specific environmental conditions or if specific habitats are available. It is important that the trajectories being evaluated are heritable to some extent and their expression is influenced by environmental conditions. The loss of key habitats for certain life-history trajectories may limit the expression of traits associated with that trajectory. Effects related to the time of sampling complicate monitoring life-history diversity. Only through extensive monitoring of key life-history traits (at multiple life-history stages and over several years) can a useful measure of diversity be obtained. It is envisioned that critical life-history data could be obtained in conjunction with juvenile and adult abundance monitoring, and that representative basins might be selected within major life-history/ecological strata.

While there is limited historical information available to establish life-history diversity benchmarks, it is reasonable to assume that the life-history diversity is strongly correlated with the ecological diversity of habitats utilized by populations. Therefore, the restoration of historical life-history diversity may be best achieved through the restoration of historical habitat diversity. Which elements of historical habitat were critical in maintaining life-history diversity remains to be determined. Historical (baseline) life-history diversity could be derived from the analysis of salmonid populations in pristine habitats, particularly those from similar ecological zones.

Approach Selected

In establishing specific recovery criteria goals, and actions to achieve those goals, it is necessary to identify which life-history trajectories are critical to population viability and how one reestablishes the expression of those trajectories. This requires understanding the suite of life-history trajectories that are possible and how they relate to specific environmental conditions. Where possible, one should identify the current and likely historical life-history characteristics of naturally produced fish within the target population(s) and identify those life-history traits and trajectories that are important to population viability. Local recovery entities should focus on life-history/habitat relationships within their watersheds: where this is not possible, information from other populations within the same life-history/ecological zone stratum should be relevant. In the absence of specific information, management and recovery actions must be based on presumed life-history characteristics. This information will be critical for identifying which life-history strategies historically contributed and are currently contributing to the persistence of a population. Estimates of historical levels of diversity provide an important benchmark for population viability. Historical life-history levels may provide the only proven template for viable populations, although there may be some fraction of historical levels that is sufficient for viable populations. Local recovery entities must consider management actions that can target those habitats that existing strategies rely upon, or actions that restore life-history

strategies that are no longer present. Whether the life-history trajectories are sufficiently diverse is ultimately expressed in a population's sustainability.

Within-Population Diversity Recovery Strategies

This document has focused on defining the criteria that describe a viable population; however, among the population attributes within-population diversity is somewhat different from the others. The diversity characteristics measured reflect adaptation to local environmental conditions, and viability criteria may well differ between populations and certainly between strata. Environmental variability may produce substantial annual variation in character expression, but without any change in population viability. Finally, the only valid measure for evaluating levels of diversity is by monitoring the fitness of subsequent generations. For these reasons, recovery strategies should not focus on producing specific characteristic levels (i.e., specific percentages of yearling migrants), but reestablishing those processes that will result in the expression of appropriate levels of diversity.

1. Maintain or restore conditions that allow for the local adaptation of naturally produced populations.

Historically, established populations were adapted to local habitat conditions. Life-history trajectories evolved over generations of natural selection. If populations have been extirpated or modified through artificial propagation activities, harvest, or habitat degradation, the genetic basis for historical life-history diversity is also lost or modified. In most of these cases, the conditions to which the fish needed to adapt changed at a rate that was too rapid for evolutionary processes to function effectively. Efforts to reestablish locally adapted populations will most likely benefit from the use of founding populations similar to the historical populations. Founding populations may include hatchery populations that may have assimilated elements of the historical population, or geographically proximate populations from ecologically similar basins. The recovery of life-history diversity in existing or reestablished populations will depend on these populations' ability to adapt to local conditions. The pace of local adaptation depends on habitat conditions, the degree of similarity between the historical and founding populations, the continued level of interbreeding between the local population and nonadapted individuals from local or distant hatchery programs or populations, and the intensity of natural selection. While it is not possible to set criteria for the process of local adaptation, it is possible to establish criteria for conditions that facilitate this process. Specifically, one could limit the level of hatchery-origin fish contributing to a naturally spawning population. This permissible level might depend on the genetic similarity between the hatchery and naturally spawning populations. Monitoring biochemical genetic markers in populations is a useful measure of the degree of a population's reproductive isolation and the relative degree of relatedness of different populations. Similarly, harvest regimes can selectively affect portions of a run and should be modified where necessary. It is likely that this process will occur concurrently with habitat restoration efforts. Where the native population has been extirpated or heavily influenced by introduced stocks, judicious selection of a founding population and the effective restoration of local adaptation processes will greatly facilitate recovery efforts.

2. Maintain and restore key historical habitats, with the expectation that populations will exploit ecological opportunities if these areas are recovered through actions such as:

- a. restoration of access to historical habitat (e.g., side-channel and headwater areas) through modification or removal of nonnatural barriers;
- b. restoration of important hydrological processes.

3. Maintain and restore “important” historical life-history trajectories and take actions to establish/restore the necessary habitat pathways for those trajectories.

The expression of specific life-history traits is predicated on the presence of specific habitats. Therefore, it is unlikely that historical life-history diversity can be restored without the restoration of historical habitat, or, at a minimum, those habitat elements that are critical to the expression of life-history trajectories. Where information on historical life-history traits is limited or absent, the restoration of historical habitats may provide a useful surrogate for restoring diversity. Local recovery entities will need to identify distinct habitats that were historically occupied and/or historical life-history trajectories. Where basin-specific information is lacking, it may be necessary to infer similarities between ecologically similar basins. Based on the information available, local restoration efforts can be either life-history or habitat driven.

Critical Uncertainties

When establishing criteria for within-population diversity, there is considerable uncertainty in defining how much life-history diversity is enough to sustain a population at VSP levels. Similarly, there is little information available to establish how much habitat diversity is needed to maintain the necessary level of within-population diversity. Historical life-history traits and trajectories provide us with the template most likely to support viable populations. An examination of historical and current information on life-history traits and historical and existing habitat conditions may provide the necessary insight to identify those life-history trajectories that are fundamental to population diversity and sustainability. Historical information on most populations is sparse, and an understanding of the relationships between life-history traits and habitats is still being developed. In many cases, recovery actions could focus on the restoration of historical habitats, with the assumption that existing populations have retained the appropriate genes necessary to exploit these habitats. It may take some time before managers are able to identify the key habitats necessary to restore life-history trajectories. Alternatively, if a population has gone through a genetic bottleneck it may not be able to express certain trajectories. Under these conditions, actions by local managers may be restricted to recovery efforts with very generalized goals for diversity criteria. It is likely that populations within the same life-history/ecoregion strata would share similar life-history criteria. As monitoring programs develop and our understanding of biological systems improves, more specific criteria will be developed.

It is not possible to definitively identify the spectrum of life-history trajectories necessary to provide enough diversity for population sustainability and long-term evolution. Historical information is useful, but often too limited to assist in prioritizing life-history or habitat types.

Criteria based solely on historical characteristics would ignore the dynamic nature of salmon populations, salmon habitat, and environmental conditions. Habitat restoration activities may also affect the expression of life-history traits. In fact, the responsiveness of life-history traits to environmental fluctuations could be a useful measure of adequate diversity. Life-history diversity, like salmon habitat, is dynamic. Furthermore, there are probably a number of different permutations of life-history trajectories that will ensure a population's sustainability. Recovery criteria need to reflect the functional aspects of life-history diversity, exploitation of multiple habitats, and buffering against environmental variability.

Monitoring and Evaluation

The design and implementation of a monitoring program for life-history diversity within populations is daunting. First, any monitoring effort would have to be tailored to the specific criteria metric established. There are a large number of life-history traits that can be monitored at an equally large number of geographic locations at various points in a fish's life cycle. Monitoring is somewhat facilitated by the association of multiple traits within a life-history trajectory and the relationship between life-history trajectories and specific habitats. Sampling a more limited suite of traits closely related to major life stages may be sufficient. For example:

- smoltification—age at downstream emigration, marine entry (from direct observation or inferred from adult scales);
- age at maturation—from adult scales or juvenile marks (e.g., CWT, otolith marks, etc.);
- run and spawn timing—direct observation of marked fish captured in terminal fisheries or adults observed on the spawning ground.

Furthermore, monitoring traits at these life-history stages could be coordinated with juvenile and adult abundance monitoring. In some cases, monitoring the same trait through different means may provide useful information about the contribution of different trajectories. For example, the age at outmigration can be measured by monitoring juveniles as they leave freshwater systems, or scale patterns from returning adults can be examined to estimate the time of seawater entry. Both types of information are needed to estimate the relative contribution of different emigration strategies. Furthermore, monitoring over an extended time period (years) should demonstrate that a population is capable of expressing a variety of life-history types. Monitoring key life-history traits provides one measure of a population's diversity; alternative methods might include quantifying the diversity of habitat utilized or the responsiveness (degree of change in life-history trajectory composition) to environmental variation. Alternatively, EDT analysis (Mobrand Biometrics Inc., 1996) provides a method of estimating the historical and present life-history trajectories based on habitat conditions. EDT examines the relationship between life-history trajectories and the habitats on which they depend. By evaluating the quality and quantity of the habitats necessary to complete any one pathway under existing and pristine conditions, one can estimate the relative proportion of potential life-history strategies that are currently occupied. Additionally, it may be possible to identify which habitats limit life-history diversity. For this reason EDT analysis may be useful in identifying recovery actions in a watershed. There was some concern, however, that EDT only verifies the existence of habitat pathways for life-

history trajectories and does not actually verify that fish are utilizing those pathways. Furthermore, EDT life-history trajectories are based on estimated movements spatially and temporally through the watershed (with potentially hundreds of permutations), rather than considering major life-history strategies.

An alternative approach to monitoring life-history diversity would combine elements of spatial structure and juvenile/adult monitoring. The presence of fish in specific habitats can be used to infer that the life-history traits necessary to exploit that habitat are present. This may be especially true when fish are observed in newly restored or reconnected habitats.

Extensive monitoring might not be necessary for all populations within an ESU. Sampling a limited number of populations within each life-history/ecozone stratum could be sufficient to characterize the diversity for each stratum. Furthermore, if this extensive monitoring of a selected collection of populations were combined with less-intensive monitoring or the monitoring of correlated characteristics (i.e., habitat diversity as a proxy for life-history diversity) for the other populations in the strata, it may be possible to evaluate the status of life-history diversity throughout the ESU.

In addition to life-history monitoring, both naturally spawning populations and hatchery populations should be regularly monitored for genetic variation using allozymes and/or DNA analysis. While there are a number of monitoring programs currently under way, these activities should be expanded to include all relevant populations and coordinated to maximize efficiency.

Risk Characterization

The approach we have taken to integrating information from all population attributes requires that populations be characterized on a 0–4 scale for all attributes. Within-population diversity is difficult to quantify because there is a vast suite of life-history traits that can be monitored, in addition to numerous less-direct measures of diversity (for example, habitat utilization, biochemical measures of genetic variability, and population effective size). In addition to direct and indirect measures of diversity, consideration should be given to the processes involved in local adaptation (which maintain and restore appropriate diversity to populations). Of these processes, considerable information is available on gene flow. The migration of adult fish across population boundaries, whether of natural or hatchery origin, reduces the efficiency of local adaptation. Within a population, the presence of relatively large numbers of hatchery-origin fish, especially those of nonlocal origins, within a population should be scored as having negative diversity consequences. Other anthropogenic factors (harvest, hydropower flow programs, timing of irrigation withdrawals, etc.) also affect life-history diversity and should be considered in evaluating the persistence category for diversity. Based on the guidelines presented in this section, the following system will be employed until the relationship between specific life-history traits and population viability is better understood and can be more quantitatively approached. Information related to within-population diversity can be assigned to one of three equally important attribute elements: (1) life-history phenotypes, (2) genetic variability or effective population size measures, and (3) local adaptation processes. Professional judgment will be used to evaluate each element relative to the presumed historical levels for that species and population. Similarly, it is necessary to employ professional judgment to weigh the importance of life-history traits being monitored (directly or indirectly) in relation to the ecological conditions that the population experiences. As discussed earlier, the expression

of variation in life-history traits depends on the ecological opportunities that exist at any point in time. Any evaluation should include observed changes in life-history traits over time in response to environmental fluctuations. The ability of a population to respond to environmental perturbations provides a definitive measure of adequate levels of diversity. Where a population's response to such perturbations has not been observed, more reliance needs to be placed on genetic variation/effective population and local adaptation element evaluations.

We expect considerable refinement of the risk characterization to occur in the future, especially with respect to the quantitative levels. Given the existing status of listed ESUs in the WLC recovery domain, it is unlikely that the component populations would achieve VSP persistence levels for some time. Improvements in the quantity and quality of monitoring over the next few years will provide the necessary information to develop a more quantitative approach to risk categorization. In the near term, risk categorization will be most useful in helping recovery entities focus their activities on the limiting components of population attributes.

Table 6.1 Risk characterization for within-population diversity.^a

Population Persistence Category	Within-Population Diversity
0	All four diversity elements ^b are well below predicted historical levels, extirpated populations, or remnant populations of unknown lineage.
1	At least two diversity elements are well below predicted historical levels. Population may not have adequate diversity to buffer the population against relatively minor environmental changes or utilize diverse habitats. Loss of several major presumed life-history phenotypes is evident; genetic estimates indicate major loss in genetic variation and/or small effective population size. Factors that severely limit the potential for local adaptation are present.
2	At least one diversity element is well below predicted historical levels; population diversity may not be adequate to buffer strong environmental variation and/or utilize available diverse habitats. Loss of life-history phenotypes, especially among important life-history traits, and/or reduction in genetic variation is evident. Factors that limit the potential for local adaptation are present.
3	Diversity elements are not at predicted historical levels, but are at levels able to maintain a population. Minor shifts in proportions of historical life-history variants, and/or genetic estimates, indicate some loss in variation (e.g., number of alleles and heterozygosity), and conditions for local adaptation processes are present.
4	All four diversity elements are similar to predicted historical levels. A suite of life-history variants, appropriate levels of genetic variation, and conditions for local adaptation processes are present.

^a Guidelines for incorporating uncertainty due to incomplete or poor data quality are presented in Section 3 and Appendix C.

^b The four diversity elements of the criterion are (1) life-history diversity, (2) gene flow and genetic diversity, (3) utilization of diverse habitats, and (4) resilience and adaptation to environmental fluctuations.

7. HABITAT CRITERIA

GENERAL HABITAT CRITERIA GUIDELINES

1. **The spatial distribution and productive capacity** of freshwater, estuarine, and marine habitats should be sufficient to maintain viable populations identified for recovery.
2. **The diversity of habitats** for recovered populations should resemble historical conditions given expected natural disturbance regimes (e.g., wildfire, flood, volcanic eruptions, etc.). Historical conditions represent a reasonable template for a viable population; the closer the habitat resembles the historical diversity, the greater the confidence in its ability to support viable populations.
3. **At a large scale, habitats should be protected and restored, with a trend toward an appropriate range of attributes for salmonid viability.** Freshwater, estuarine, and marine habitat attributes should be maintained in a nondeteriorating state.

Overview

Habitat, as used here, comprises the myriad environmental components and processes operating over time and space that affect the growth, behavior, distribution, and survival of individual salmonids and therefore the viability of salmonid populations. Delisting criteria will need to address the freshwater and estuarine habitat characteristics necessary for persistence and recovery of a species. The importance of habitat is one of the primary purposes of the ESA [Section 2(b)]: “to provide a means whereby the ecosystems upon which threatened and endangered species depend may be conserved.” Habitat criteria help ensure that recovery is not ephemeral. Inherent time lags occur between certain types of habitat modification and fish population response. For example, the negative effects of removing a riparian buffer may only be observed during infrequent extreme weather events. The dynamics of a fish population in the interval between weather events may suggest that it is viable. However, a broader evaluation, one which included habitat status, might reach a different conclusion about long-term risks facing the population. Thus, evaluating habitat provides unique information about a population’s status. A delisting decision based solely on a fish population’s performance may be short-sighted if the fundamental habitat problems that led to the initial listing are not solved or if new habitat degradation is not prevented.

Habitat criteria also provide a check on viability criteria developed through demographic models. A key assumption of the demographic modeling used to develop biological viability criteria is stationarity, the assumption that the behavior of the population over the observation period will continue into the future. Projecting the future dynamics of relevant habitat features with long temporal responses, such as river sediment dynamics, can provide an important check on the validity of the stationarity assumptions. Habitat criteria have the potential to provide an additional predictive element, which augments our ability to provide for salmonid persistence and recovery into the future. Habitat criteria will help to prevent a delisting decision based on demographic models for which assumptions are no longer valid.

Developing habitat criteria will involve technical analyses that are, in many ways, more complex than the ones needed to develop the criteria that describe salmon population attributes. The challenge will be to determine the habitat conditions the listed salmon species needs to persist throughout its full range and all its life stages. As recovery planning progresses to move beyond the already challenging task of defining viable habitat characteristics, it will be necessary to describe the processes that have led to their deterioration. An additional challenge facing the development of habitat criteria is identifying appropriate temporal and spatial scales. The watershed, including the estuary, is the basic ecological restoration unit of the WLC domain. Land- and water-use practices within watersheds are intimately tied to the condition of the streams or rivers that drain them. Habitat conditions in the mid and upper Columbia River watersheds will impact the estuarine and nearshore conditions experienced by listed fish in the WLC domain. Habitat criteria will necessarily consider entire watersheds within and related to the WLC domain.

Strategies Selected

For all WLC listed salmon ESUs, habitat loss and destruction was identified at the time of listing as one of the major five factors for decline. VSPs require an abundance of high-quality habitat, distributed throughout entire watersheds, including freshwater and estuarine areas, in a manner that will support all life-history stages and provide for connectivity among the various life stages. In this sense, habitat is not just another one of the “H’s”—salmon populations do not need hydroelectric dams, hatcheries, or harvest to be viable but they do need good habitat. Habitat conservation, restoration, and protection are essential components of salmon recovery.

While the TRT has focused thus far on criteria related to salmonid growth and abundance, we have begun to conceptually frame some issues related to freshwater and estuarine habitat criteria. This work is in the preliminary stage, but has resulted in general habitat criteria and a preliminary list of habitat attributes to be considered (Table 7.1). These criteria largely serve as placeholders until more specific criteria can be developed. TRT members have explored some specific habitat criteria involving temperature tolerances for salmonids (Appendix L). These temperature criteria were developed as a first example of what specific habitat criteria might look like. Temperature was selected for this first example not because it was identified as the most important factor, but because it is likely to be one of the important factors and data availability made it a tractable place to start. Additional questions about temporal and spatial scale still have to be answered.

The habitat criteria were derived from well-established principles about the relationship between salmon and the habitats on which they depend. First, habitats have a limited carrying capacity, which depends on habitat quality. In order to recover populations, there must be adequate amounts of freshwater and estuarine habitat of sufficient quality to support viable populations identified for recovery. Salmon require different types of habitat at different life-history stages (Table 7.1), therefore the habitats on which these populations depend must also have a spatial distribution sufficient to maintain viable populations identified for recovery. Second, salmon have evolved under natural disturbance regimes. These disturbance regimes have resulted in diversity, over time and space, of habitat types and conditions. Therefore, the diversity of habitats for recovered populations should resemble historical conditions given expected natural disturbance regimes (e.g., wildfire, flood, volcanic eruptions, etc.). Historical

Table 7.1 Examples of habitat attributes and life-history stages for which each attribute is particularly critical.

Habitat Attribute	Return to Spawn	Egg Incubation	Juvenile Rearing	Out- migration	Ocean Transition
Stream flow	X	X	X	X	X
Water temperature	X	X	X	X	X
Sediment		X			X
Dissolved oxygen		X			X
Chemical contaminants	X	X	X	X	X
Nutrients		X	X	X	
Physical barriers (fish passage)	X		X	X	
Percent pools			X		
Large woody debris			X		
Substrate	X	X			
Off-channel refugia			X	X	
Interactions with exotics	X		X	X	X
Streambank stability		X	X	X	
Trophic dynamics	X		X	X	X
Floodplain connectivity			X	X	X
Tidal flat connectivity				X	X
Channel width:depth ratio	X	X	X		
Road density		X			
Landscape disturbance history					
Riparian condition	X	X	X	X	

conditions represent the only known template for a viable population, and the closer the habitat resembles the historical diversity, the greater the confidence in its viability. Third, large-scale processes have a driving influence on both habitat conditions and salmonid population performance. At smaller scales, habitat quality varies naturally. Therefore habitats must be maintained and restored to ensure a trend toward an appropriate range of conditions for salmonid viability. At a large scale, freshwater and estuarine habitat conditions should be maintained in a nondeteriorating state.

Relationship of Habitat to Other Criteria

Since habitat is a key driver of fish population performance, we expect some overlap between criteria describing habitat and the other criteria (i.e., growth and abundance, JOM, diversity, and spatial structure). However, as described in the introduction to this document, each criterion contributes distinct information about population risk. The greatest potential overlap is between spatial structure and habitat criteria. An alternative conceptual structure for criteria development would place spatial structure, habitat quality, and habitat dynamics as components of the habitat criteria. The TRT elected to retain spatial structure as a separate attribute because

evaluating the spatial structure of a population involves examining the distribution and dispersal behavior of the fish themselves. This is a different exercise than examining the distribution and quality of available habitats. Evaluating habitat spatial structure and quality allows for the incorporation of sublethal effects of reduced habitat quality and of a gradation in habitat quality beyond the presence or absence of fish. The high level of correlation between the habitat and spatial structure criteria should be considered as the criteria are integrated into an overall assessment of population viability.

Critical Uncertainties

In developing habitat criteria, several assumptions must be acknowledged. First, distribution of historical populations is assumed to be reasonably accurate. Second, the habitats in which historical populations resided are assumed to have been spatially arranged and of sufficient productive capacity to provide demographically independent VSPs. Third, the ideal arrangement and spatial distribution of habitats for a given viable population is unknown and likely fluctuates over time. Fourth, our understanding of fish-habitat relationships, while improving over the last several decades, is incomplete, making it difficult to accurately predict productive capacities for specific watersheds. Fifth, our understanding of natural disturbance regimes and the roles they play in the ecological succession of aquatic habitats for salmon is imperfect. Incorporating these concepts into “recovered salmon landscapes” in a manner that connects headwater areas to mainstem rivers to estuarine habitats must rely heavily on analyses generated from historical habitat reconstructions and simulations of natural disturbance regimes tempered with the existing, but limited, empirical data and professional judgment.

Future Development of Specific Habitat Criteria

The habitat criteria need to be developed into more specific “measurable and objective criteria” to evaluate extinction risk. Two general approaches could be taken for future development of habitat criteria. One approach is to develop standards for specific habitat attributes, then evaluate all habitats relative to those standards. An example of this approach is the maximum temperature criteria developed in Appendix L. An alternative approach does not rely on the initial development of uniform standards; instead it focuses on watershed analyses to identify a distribution of target conditions appropriate for a particular time and place. The distribution of target conditions would be determined by examining historical and current watershed processes and correlating them with the requirements of fish populations. Both approaches might ultimately result in similar criteria. That is, the standards approach might result in the development of condition-dependent standards that can be tailored to a specific time and place, while the watershed analysis approach might result in the identification of generally required habitat conditions that could be developed into a set of standards for specific locations. We are not providing a full discussion of the strengths and weaknesses of the two approaches in this document, but both have merit and the TRT has been divided on which approach to pursue. In addition, the TRT has not identified the best methods for incorporating nearshore conditions

into the habitat criteria, but the members do agree that further work is needed on nearshore habitat requirements and criteria.

Habitat Risk Characterization

Because they are so numerous and complex, the environmental factors known to affect salmonids cannot be readily comprehended, inventoried, and evaluated. To facilitate the characterization of habitat and its ability to support VSPs, we rely on a subset of key habitat components or processes, measured at appropriate times and places, as indicators of habitat quality. These key habitat indicators, when individually and collectively monitored and evaluated, provide useful information of the current status and trend of environmental conditions. Based on measurements of indicators and our understanding or observation of their effects on fish, we can make some judgment as to the overall quality of habitat present at a particular location and time. Moreover, if we are able to reliably forecast future changes in habitat conditions based on current trends, we can also predict their probable impact on salmonid populations.

Indicators of important habitat components and processes have been compiled in the scientific literature, and are routinely used by fisheries scientists to quantitatively describe habitat conditions. Table 7.1 lists a subset of habitat indicators derived substantially from the Matrix of Pathways and Indicators developed by NMFS as part of its “Checklist for Documenting Environmental Baseline and Effects of Proposed Action(s) on Relevant Indicators” (NMFS 1996). A more extensive list of habitat indicators, including known or presumed effects on the survival of different life stages of salmonids, are used to parameterize the Ecosystem Diagnosis and Treatment model (Moberg Biometrics 1996).

One or more specific parameters and evaluation protocols can be identified for each indicator. For example, streamflow at a specific location can be statistically described by reference to several hydrologic parameters that characterize the magnitude, frequency, timing, duration, and rate of change in discharge for the period of record.

The actual parameters measured for each indicator are referred to as indicator variables. The biological response of individual fish to different levels of indicator variables or, alternatively, the quantitative relationship between a given habitat component or process and the associated fish population, can be deduced or postulated based on empirical data or expert judgment. These types of relationships vary with species, life stage, and interactions with other variables.

There is ample evidence that fish respond in predictable ways to changing habitat conditions. In particular, we know that certain conditions are associated with and presumably are responsible for different levels of fish performance. Habitat conditions are acceptable—that is, capable of supporting viable populations of salmonids—if they fall within the normal range of variability for undisturbed systems.

For each indicator variable, there is a threshold below which conditions are considered unacceptable or impaired. This impairment threshold is a value or range of values that are specific to the species, life stage, and area (i.e., habitat type, watershed, ecosystem) in question. In areas where values fall below identified impairment thresholds for one or more key indicator variables, the habitat is considered impaired.

The indicators, indicator variables, and impairment thresholds (e.g., “properly functioning [habitat] conditions”) recommended by NMFS (1996, 1999), Spence et al. (1996), and other sources reported in the literature are a useful starting point for assessing habitat within areas utilized by salmonid populations in freshwater, estuarine, and marine ecosystems. The timing, distribution, and specific ecological requirements of the population and life stage in question will need to be determined at appropriate spatial scales. The relationship between habitat and fish performance, and the range of values associated with impaired and nonimpaired habitat, as measured by ecologically relevant indicator variables, will need to be determined through best available science applied to local conditions.

Separate lists of key habitat features can be compiled for freshwater, mainstem, estuarine, and marine areas occupied by the population over its life cycle. If necessary, more refined lists of indicators and indicator variables can be compiled for different life stages and applied to subareas within these larger ecosystems.

Once appropriate evaluation metrics and standards have been defined, the habitat within a particular ecosystem (e.g., freshwater, main stem, etc.) can be qualitatively evaluated with respect to its overall effect on population performance. Both existing and future habitat conditions may be predicted. Those responsible for the assessment will need to devise a rational approach to measuring or estimating, weighting, and integrating the values obtained for different indicator variables and indicators. Careful consideration will need to be given to the selection of indicator variables, the spatial and temporal scales at which they are measured and expressed, and their synthesis and integration across appropriate scales of time and space, as defined by the population.

The result of the habitat assessment, which necessarily will rely heavily on expert opinion, will be a characterization of habitat within the occupied (or potentially occupied) geographic range of the population for the specified time period of interest. The assessment comprises two types of information: (1) a measure of the current status of the habitat within areas occupied by the population; and (2) a measure of the likely trend in habitat conditions, given anticipated natural and human-induced changes in the environment over the foreseeable future. By “future” we mean over a 20- to 100-year time span. Habitat within areas occupied by each population can be categorized into one of five levels as described in Table 7.2.

The recommended procedure for evaluating habitat considers the existing and probable future status of habitat measured at appropriate scales, relative to impairment thresholds specified for key environmental indicator variables. Probable future conditions can be predicted based on observable trends in habitat condition, as indexed by indicators of human population growth and distribution, land and water use, and other predictors of natural and human-caused environmental impact.

Table 7.2 Risk Characterization for habitat criteria.

Persistence Probability Category	Description
0	Habitat is incapable of supporting fish or is likely to be incapable of supporting fish in the foreseeable future.
1	Habitat exhibits a combination of current impairment and likely future conditions such that the population is at high risk of extinction.
2	Habitat exhibits a combination of current impairment and likely future condition such that the population has a moderate risk of extinction.
3	Habitat is nonimpaired and likely future conditions will support a viable salmon population.
4	Habitat conditions and likely future conditions support a population with an extinction risk lower than that defined by a viable salmon population. Habitat conditions consistent with this category are likely comparable to those that historically existed.

8. WITHIN-POPULATION SPATIAL STRUCTURE CRITERIA

WITHIN-POPULATION SPATIAL STRUCTURE CRITERIA GUIDELINES

1. The spatial structure of a population must support the population at the desired productivity, abundance, and diversity levels through short-term environmental perturbations, longer-term environmental oscillations, and natural patterns of disturbance regimes. The metrics and benchmarks for evaluating the adequacy of a population's spatial structure should specifically address:
 - a. Quantity: Spatial structure should be large enough to support growth and abundance, and diversity criteria.
 - b. Quality: Habitat underlying spatial structure should be within specified habitat quality limits for life-history activities (spawning, rearing, migration, or a combination) taking place within the patches.
 - c. Connectivity: Spatial structure should have permanent or appropriate seasonal connectivity to allow adequate migration between spawning, rearing, and migration patches.
 - d. Dynamics: The spatial structure should not deteriorate in its ability to support the population. The processes creating spatial structure are dynamic, so structure will be created and destroyed, but the rate of loss should not exceed the rate of creation over time.
 - e. Catastrophic Risk: The spatial structure should be geographically distributed in such a way as to minimize the probability of a significant portion of the structure being lost due to a single catastrophic event, either anthropogenic or natural.

Overview

The spatial distribution of a population is the distribution of fish and the processes that generate that distribution (McElhany et al. 2000). Spatial distribution is important because the viability of a population is closely linked to it in both the short and long term. Viability is linked to spatial structure in a variety of ways. Most simply, there must be sufficient high-quality habitat to support the population, and habitat areas must be connected so that fish can move from one area to the next as their life history requires. However, spatial structure affects viability in more subtle ways as well. We know from metapopulation theory that spatial structure can affect extinction risk in ways that would be undetectable from short-term observations of population growth and abundance (Hanski and Gilpin 1997, Tilman and Lehman 1997, Cooper and Mangel 1999). Some areas of high abundance may actually be production sinks, while other areas with less fish abundance may be responsible for most of the production (Pulliam 1988). Also, because of the demographic interplay between areas, the genetically effective size of a population may be

smaller than would be otherwise assumed (Whitlock and Barton 1997). Spatial structure is also linked to genetic diversity (see Section 6). Diversity in population structure promotes genetic diversity, which is a key component of long-term viability. Finally, the spatial structure of a population often determines its susceptibility to catastrophic risk. In general, the more dispersed the population, the less likely it is to be heavily impacted by a landslide, volcanic eruption, or other catastrophe. An extensive account of spatial structure and its role in salmonid population viability can be found in McElhany et al. (2000).

Spatial structure can be considered at any spatial scale. Spatial structure at the ESU level has already been covered in an earlier section. In this section we consider the spatial structure of individual demographically independent populations identified in the Lower Columbia and Willamette ESUs (Myers et al. 2002). The spatial structure of an individual population is characterized by the spatial distribution of its habitat, its dispersal patterns and dynamics, and the quality and quantity of its habitat. Complexity of spatial structure can be expected to vary according to species; life-history types; and the basin's hydrographic, physical, chemical, and biological processes. Thus, we would expect to see differences in spatial structure between chinook salmon and steelhead populations, between spring- and fall-run chinook populations, and possibly even between fall-run chinook populations in the coastal ecoregion and those in the Cascade Crest ecoregion.

Because spatial structure cannot exist without supporting habitat, there can be a natural tendency to consider the two as synonymous. However, certain aspects of spatial structure clearly are not habitat, and aspects of habitat are not spatial structure. Figure 8.1 presents a good conceptual framework for understanding how these two aspects of viability relate to each other, though opinions may vary about the relative sizes of the circles and the extent of overlap.

The area of overlap in Figure 8.1 reflects the extent to which spatial structure is determined by habitat distribution, quality, and quantity. The area of no overlap reflects the determination of spatial structure by genetic differentiation and population-specific dispersal patterns. The large amount of overlap with habitat means that a considerable amount of discussion about spatial structure will necessarily be a discussion of habitat. At times this may

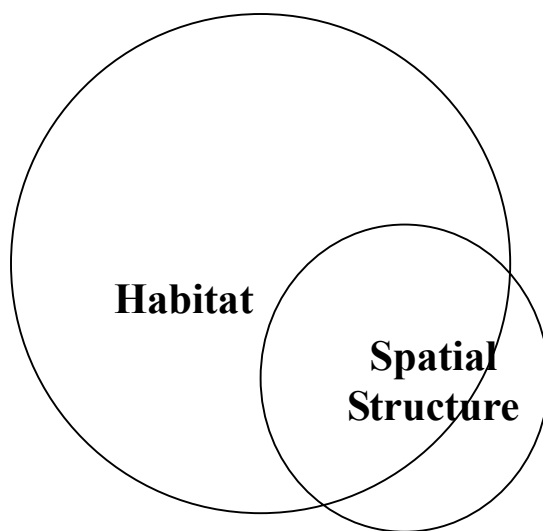


Figure 8.1 Venn diagram of relationship between habitat and spatial structure.

seem redundant. We have tried to limit this redundancy but feel a certain level of it is appropriate, even to the extent of having close correspondence between several of the spatial structure and habitat criteria guidelines.

In discussing the spatial structure criteria, it is useful to use the term patch (Levins 1969) to refer to an area of habitat. We define patch as an area of homogenous habitat that is consistently used during a particular life stage. Patches can be anywhere from a few meters to many kilometers in size. They may be continuous or discontinuous, and their boundaries may be determined by discontinuities in habitat features. However, utilization must also be considered. This can make clear identification of patches difficult. For example, consider a long stream reach that may appear to be good spawning habitat, but in which the fish spawn in only two very localized areas. The homogeneity of the reach may be a strong argument for considering it a single patch for spatial structure assessment purposes, but the utilization pattern would argue that this area consists of a group of occupied and unoccupied patches. However, in a year in which the number of returning adults is very high, spawners may utilize the entire reach, making it in effect a single patch. We classify patches according to their most obvious usage—spawning, rearing, or migration—but these distinctions are somewhat arbitrary. For example, spawning patches are also invariably early rearing patches, and because fish move through them on their way to other rearing areas, they are also to some extent migration patches.

Approaches Considered

Our initial goal was to develop a simple and consistent set of criteria, consistent with the five general recommendations of McElhany et al. (2000), that could be applied uniformly over all populations of all species without creating situations that seem either arbitrary or illogical. The simplest approach was to require a minimal level of dispersal, for example, requiring that spawning occur in at least two or three geographically distinct areas. Obvious variation among species made this approach seem illogical. For instance, Wind River summer steelhead now spawn and probably historically spawned in the headwaters of Panther Creek, Trout Creek, and the Wind River, whereas Coweeman River fall chinook salmon now spawn and probably historically spawned in a single long stretch of the main stem of the river. Thus, a simple dispersal rule makes sense for the steelhead, which exhibit a considerably branched (dendritic) spawner distribution, but not necessarily for the fall chinook salmon. As we examined a variety of existing and presumed historical spatial structures, it became clear that geological features also influenced spatial structure complexity. Some basins have the capacity to support a highly dendritic structure, while others did not. It became clear early on, as with the diversity criteria (Section 6) that the only logical, uniform approach was a set of general criteria flexible enough to allow for variation among populations and basins.

In the face of uncertainty, the general guideline on all viability issues is to use the historical situation as a template (McElhany et al. 2000). We considered this an appealing concept because our knowledge of habitat processes, and fish-habitat and dispersion dynamics, is too limited to specify appropriate spatial structures for individual basins. Presumably the historical spatial structure worked, so it should provide a good model. But there are problems with wholesale adoption of the historical spatial structure as a recovery template. First and most obvious, the historical spatial structure may not be well known and may even be unknown.

Second, the spatial structure within natal watersheds of most populations within the Lower Columbia and Upper Willamette ESUs has been changed substantially by land use, channel modification, and hydroelectric development; and the downstream structure of all populations has been changed by development of the Columbia. Developing appropriate spatial structures to support viable populations in these ESUs in many cases is not a matter of repairing habitat here and there to restore the historical structure. Some of these basins may have changed so much that any workable spatial structure would be substantially different from the historical spatial structure. Finally, spatial structure is not an end unto itself. An appropriate spatial structure is an interaction between fish and habitat that supports the population in ways that allow it to be both abundant and resilient. An appropriate spatial structure is thus one that works, and that may or may not require it to be quite similar to the historical structure. Any recovery approach will almost certainly require restoration of parts of the historical spatial structure, but devoting ourselves too slavishly to the complete restoration of the historical structure could be an expensive, misguided effort. We concluded that developing a viable spatial structure should be informed by knowledge both of the historical spatial structure and of the spatial structures of relatively undisturbed conspecific populations in comparable settings. However, following this approach, the point of attempting to completely restore the historical spatial structure or mimic that of a reference population may not be necessary or wise. Functionality is the key.

Assuming that a historical or reference population spatial structure will be used to guide development of a viable spatial structure, we then considered simple utilization/dispersion criteria based on these templates. Possible criteria were that spawning (or other usage) should occur in all tributaries that historically supported spawning or in some fraction of those tributaries. Several problems arose, including defining what percentage of historical was sufficient, and how to take into account areas in which historical spawning areas seemed to be permanently lost. We also explored the idea of core areas, regions of very high utilization by one or more life stages. Again, there were several problems. Any rule that allowed some historical spots to be neglected ran the risk of having a core area neglected. Also, it wasn't clear that trying to force a former core area to become one once again always made sense. Spatial structure is dynamic, because the forces creating it are dynamic. For example, natural processes are constantly altering stream characteristics such as spawning substrate distribution and depth (Benda 1994, Reeves et al. 1995). In a properly functioning basin, natural processes will destroy habitat patches and create new ones. The relative importance of patches will be dynamic as well, with sources becoming sinks and vice versa. A healthy system will contain a diversity of patch quality and occupancy, as fish move from degrading patches into new high-quality ones. Core areas will come and go. The key to viability is insuring that the spatial structure processes create a dynamic structure that is always adequate to maintain the population. Trying to force a particular usage or dispersal pattern could easily be a misguided effort. Again, the goal is functionality.

In developing a viable spatial structure, existing structure needs to be considered. What structure there is should be conserved and used as a foundation in developing the recovery spatial structure. Special attention should be given to protecting existing core usage areas. Study of these areas and other currently used areas will be invaluable in understanding population-specific interactions of fish and habitat.

A key characteristic of spatial structure is connectivity. Adults must be able to get to the spawning grounds, fry must be able to get from spawning areas to rearing areas, and outmigrants must be able to go downstream to the Columbia River. Connectivity does not have to be

permanent, but needs to be seasonally adequate; that is, when fish need to move from one area to another, the areas need to be connected. In restoring connectivity, we should consider that existing connectivity may give us an erroneous picture of historical spatial structure. What may appear to be a connectivity problem for one species may limit the life-history diversity of another. Connectivity can be achieved by natural or artificial means. Natural connectivity is preferred over artificial means, such as trapping and hauling, for two reasons. First, human interventions into the natural life cycle of the fish may cause domestication, diminishing the ability of the population to sustain itself without that intervention. Second, assisted migration is inherently risky due the vagaries of funding and planning. However, we realize that achieving natural connectivity may be very difficult and perhaps not possible in the case of large dams, such as those on the Cowlitz, Lewis, and Willamette tributaries. Also, there may be occasions in many populations when assisted migration will avoid catastrophic losses.

Spatial structure is intimately associated with genetic differentiation. Spatial structure encompassing a diversity of habitats promotes genetic differentiation. Genetic differentiation in turn allows exploitation of new habitats and thus expands spatial structure. An appropriate spatial structure should therefore promote genetic differentiation, but how this translates to physical spatial structure is unclear. Clearly, quality habitat should be distributed in a way that encourages both natural patterns of fish dispersal and gene flow. However, maintaining the appropriate size and complexity of spatial structure is a difficult balancing act. In a healthy spatial structure, some patches may be so remote from others that homing fidelity will create some level of genetic differentiation, which is desirable. However, there may be a trade-off between distance and occupancy, with closer patches tending to be more occupied (Dunham and Reiman 1999). In addition, if patches are so distant as to be near those of other demographically independent populations, natural straying may blend populations. On the other hand, if the spatial structure is too restricted, occupancy may be assured but diversity and catastrophic risk may be increased. Although it was clear that too little or too much dispersal might have undesirable consequences, we were unable to distill these general ideas into criteria that would provide any real guidance.

A special case of genetic differentiation is the existence of subpopulations. In Myers et al. 2002, subpopulations are listed for many putative historical demographically independent populations, considering that there may have been some opportunity for genetic differentiation among major tributaries within a population's spatial area. In no case, however, was there actual evidence of subpopulations with any degree of genetic distinctness. So while it seems logical and desirable that historical subpopulation structure should be part of a viable spatial structure, we have virtually no guidance on what historical subpopulation structures were. Here is another case in which reference to other conspecific but less impacted populations in similar settings would be useful. Simple rules, such as insisting the population should inhabit all the major stream areas within its basins, simply do not consider the possible source-sink dynamics underlying what might have historically existed. An especially interesting case is putative historical demographically independent populations that span multiple basins draining independently into the Columbia, such as Elochoman fall chinook. The geographical range of this population includes both the Elochoman River and the Skamokawa Creek basins. In such cases, it makes sense to require some utilization of both basins, until it is shown that it is unlikely a single stock inhabited both.

A population's spatial structure should protect it from losses due to catastrophes (see Appendix K) or normal disturbance regimes. A single hundred-year flood, landslide, fire, or other catastrophic event should not be able to destroy a significant portion of the structure.

Avoiding risk from volcanoes will be difficult and perhaps impossible in basins near Mount St. Helens, Mount Rainier, and Mount Hood. There may also be some risk in the Willamette tributaries. A number of anthropogenic factors also need to be considered, with the most important being spills of toxicants. With so much of the drainages in the habitats of these ESUs close to railroads and sizable roads, spills of toxic chemicals can be a substantial risk. It is important to recognize that all populations in these ESUs use the Columbia River estuary for migration and possibly rearing, so catastrophic occurrences in the estuary can have far reaching impacts. Knowledge of catastrophic processes and recovery processes is critical to making correct decisions about this risk to structure (Dale et al. 1998). Normal disturbance regimes obviously need to be considered as well in terms of risk to spatial structure. Viewed in one sense, they are slow catastrophes. The normal disturbance regimes need to be understood sufficiently that recovery planners believe they will result in no net loss of structure.

Finally, we also considered the total geographical extent of a population's spatial structure: in physical terms, where it starts and ends. Technically, the spatial structure of any individual population in these ESUs includes the entire area occupied by the population during its life cycle. Thus spatial structure includes distribution and habitat from spawning and rearing areas in the natal basin, downstream into major migratory corridors (the Willamette and/or Columbia), and hundreds of miles into the ocean. We recognize that in practical terms the ability of local recovery planners to understand, influence, and monitor the spatial structure of a population diminishes drastically in the downstream direction to the point of extreme impracticality. It is therefore entirely understandable that most recovery planning with respect to spatial structure will be done at the watershed level. At the same time, however, the out-of-basin freshwater and nearshore areas need to be considered as critical areas for all populations. Thus, spatial structure criteria should include consideration of these geographical areas.

Strategies Selected

As explained and implied above, no single, simple rule for spatial structure will suffice for all populations of a given species or life-history type. Spatial structures that meet recovery criteria will be population-specific, reflecting the population's characteristics and the basin's processes. We offer general criteria addressing the concerns described above and leave it to the local recovery planning entities to develop objective measurable criteria that will adequately address the general concerns outlined above.

There seems a logical way for recovery planners to proceed with developing these objective and measurable, population-specific spatial structures. Both the current spatial structure and as much as possible of the historical structure for each population must be identified and mapped. If information on historical structure is lacking, then it can be inferred from historical habitat information and from relatively unimpacted conspecific populations in similar settings. As part of these analyses, highly productive (core) spawning, rearing, and migration patches must also be identified. Immediate steps should be taken to ensure the conservation of these areas, as they will almost certainly be the foundation of the viable spatial structure to be developed. For recovery purposes, a population's spatial structure should include all freshwater and nearshore areas it utilizes. This will require that recovery planners work collaboratively on these out-of-basin areas.

Locally developed, objective, measurable criteria for spatial structure need to address the following factors:

1. **Quantity.** A population's spatial structure must be large enough to support the growth and abundance levels required for viability. In other words, the habitat base and its utilization patterns must be adequate to support the population. A population's spatial structure should be considered to include all freshwater and nearshore areas it utilizes.
2. **Quality.** The habitat areas comprising the spatial structure must be of sufficient quality, demonstrated both by habitat attributes and utilization, for the life-history activities (spawning, rearing, migration, or a combination) taking place there. Habitat quality specifications have not yet been established for all criteria we consider critical, but an example of some types of habitat factors can be found in the Matrix of Pathways and Indicators (NMFS 1996). However, as described above, distribution of habitat types and variability in habitat factors should be the goal rather than a set of static minimums.
3. **Connectivity.** The spatial structure must have permanent or appropriate seasonal connectivity, demonstrated both physically and by utilization, that allows adequate migration between spawning, rearing, and migration patches. Normal weather fluctuations may result in occasional blockage or stranding of some individuals of particular life stages, but overall connectivity should be such that significant events are rare. Additionally, consideration should be given to establishing connectivity beyond the appropriate seasonal level, because connectivity may limit the expression of life-history diversity.
4. **Dynamics.** The spatial structure must not deteriorate in its ability to support the population over time. The processes creating spatial structure are dynamic, so it will change as habitat is created and destroyed, but the rate of patch destruction must not exceed the rate of patch creation. The spatial structure, evaluated in terms of habitat attributes and fish distribution, should remain relatively constant or grow in quality, size, and complexity over a long time frame. However, in the short term an immediate change from deteriorating to nondeteriorating habitat will be needed in most basins. Protection of existing core patches should be emphasized as well as restoration of other patches. Currently unoccupied but apparently suitable patches should also be maintained, because colonization may occur over a period of years.
5. **Catastrophic Risk.** The spatial structure should be geographically distributed in such a way as to minimize the probability of a significant portion of the habitat base or the population being lost due to catastrophic events, either anthropogenic or natural (Appendix K).

Risk Characterization

The approach the TRT has taken to integrating information from all population attributes requires that populations be characterized on a 0–4 scale for all attributes. Because spatial structure is so difficult to quantify, developing a scoring system for it is difficult, and a final

scoring system has not yet been developed. We offer here a tentative scoring system that is based on three general principles.

1. The scoring system needs to consider all five spatial structure subcriteria.
2. The five subcriteria differ in temporal scope. Clearly, quantity, quality and connectivity are criteria of more immediate importance than dynamics and catastrophic risk. It is also likely that more will be known about these three subcriteria than the other two.
3. The appropriateness of the spatial structure is tied to the population size that recovery planners specify. It doesn't make sense to give a very small but stable spatial structure a high rating. The possible downside to this third consideration is that it will invariably make the spatial structure score correlated with the abundance score. However, this also makes it logical: a large population cannot exist without the spatial structure to support it.

Score	Spatial Structure
0	Spatial structure is inadequate in quantity, quality, and connectivity to support a population at all.
1	Spatial structure is adequate in quantity, quality, and connectivity to support a population far below viable size.
2	Spatial structure is adequate in quantity, quality, and connectivity to support a population of moderate but less than viable size.
3	Spatial structure is adequate in quantity, quality, and connectivity to support population of viable size, but subcriteria for dynamics and/or catastrophic risk are not met.
4	Spatial structure is adequate in quantity, quality, connectivity, dynamics, and catastrophic risk to support viable population.

This scoring system is an initial attempt, and we expect to refine it considerably, especially with respect to quantitative levels. One factor that must be considered in the scoring system is the amount of information we have on the subcriteria, and thus our confidence that the subcriteria are being met. Some discounting of scores for suboptimal information quality seems appropriate, but how to do this without additional refinement of monitoring ideas is unclear at this time (see below).

Critical Uncertainties

The viability criteria guidelines above involve numerous critical uncertainties, some of which are resolvable and some of which are not. In most cases, resolving uncertainties will require a substantial investment of time and/or resources. In this section we describe some of the major uncertainties associated with spatial structure.

Perhaps the most important uncertainty relates to our reliance on historical conditions as a recovery template. Our overall method for dealing with uncertainty in developing viability criteria, such as the diversity and ESU-wide criteria, is to attempt a restoration of normative conditions, which in this case means a return as much as possible to historical processes and spatial structure. The assumption here is that historical conditions are a reasonable template for viability on a temporal scale of 100 years or more. This assumption may be reasonable, but we often lack the data or information to describe or understand historical conditions. We simply don't know how quickly populations were created or destroyed in the past. Our observations go back about a century, and most of the old records are of commercial catch, not size and distribution of individual populations.

Our understanding of the historical processes generating habitat is also limited. We have a reasonable understanding of large-scale geological and hydrological processes in these ESUs, but there is little smaller-scale or basin-specific information. For example, although we stress the importance of recognizing the dynamics of spatial structure, we have little understanding of the rate of patch cycling generally or specifically for the basins in these ESUs. It is known, however, that patches can be stable for 15 years or more (Bilby et al. 1999). An additional complication is the disruption of historical processes by land use and impoundments, as well as our limited knowledge of how quickly natural processes can be restored after disruption. By emphasizing restoration of the natural processes that create and destroy habitats, we can avoid some of the uncertainties inherent in an engineering approach (see Beechie et al. 2002).

There is little basin-specific information on historical spatial structure. What information there is (e.g., accounts in Myers et al. 2002) focuses on known or potential spawning areas. So, although we stress the importance of historical structure as a template, we expect that in most basins this will be largely conjecture. Perhaps the best that can be done in many cases is to model potential historical patches, based on the basin's geology and presumed hydrography. In many cases, there will also be gaps in information about current spatial structure and habitat conditions, simply a consequence of too many stream miles to survey for the available number of biologists. There are undoubtedly many stream reaches of these ESUs that have not been walked by management agency biologists in years. As already mentioned, knowledge of the spatial structure of relatively unimpacted conspecific populations in similar settings may be of some use in developing insights about viable spatial structures for particular populations.

Considerable debate surrounds our ability to precisely link habitat quantity and quality with fish production. Nevertheless, a number of studies have compared fish production to habitat characterizations at various levels of scale and sophistication to modelled population-size expectations. An example of a reasonably fine-scaled approach is the ecosystem diagnosis and treatment (EDT) method (Lichatowich et al. 1995, Mobrand et al. 1997), in which habitat is rated at a number of parameters to provide reach-by-reach values for equilibrium productivity and capacity. EDT (called HPVA in this document) estimates for productivity and capacity for a number of populations in the Lower Columbia and Upper Willamette ESUs are presented in Appendix J. Another approach is the work of Holsinger (2002) determining the chinook capacity of the Stillaguamish River: habitat spawner capacity is determined by typing physical habitat, then multiplying the quantities of typed habitat by the spawner densities observed for that habitat type in the basin. How precise these and other approaches are for predicting production is unknown.

Dispersal patterns and gene flow are critical factors in determining population structure, yet our understanding of natural gene-flow rates is poor. It is clear that homing to the natal

stream is determined by both genetic and environmental components (Quinn et al. 1991), but estimates of stray rates are highly variable, even within a given species (Quinn 1993). Most of the information on stray rates comes from recovery of coded-wire tags from hatchery releases, which raises questions about imprinting and stock origins. In addition, what constitutes a stray in terms of distance from the expected return site may vary from study to study.

The above discussion is just an overview of information gaps regarding the viability criteria. Each uncertainty encompasses several subsidiary uncertainties, each one a sizable area of study in itself. Clearly we need to know far more than we do now about spatial structure and fish-habitat relationships to be able to say with confidence that a given spatial structure will properly support a population over a sustained period of time. Understanding will come only from additional studies, especially of populations where historical processes still prevail, and from long-term monitoring. An adequate monitoring plan will include determining fish numbers and distributions at all life stages and measuring a large suite of habitat parameters. Monitoring spatial structure does not necessarily need to be continuous, but it does need to span extremes of variation in fish numbers and climatic conditions for changes in patch quality and occupancy to be evident. The duration of intensive monitoring should also be long enough to achieve an understanding of patch colonization, especially when evaluating restored or constructed patches. Wherever possible, monitoring should be linked to tests of specific hypotheses about population dynamics or fish-habitat relationships.

Monitoring and Evaluation

Monitoring spatial structure is conceptually simple within a given time stratum. There are two basic elements:

1. distribution and quantities of fish by life stage,
2. measurements of key habitat parameters in the areas where fish are present or desired to be present.

The information gathered in multiple time strata then must be compared to determine trends in either utilization or habitat quality. The same information will be used qualitatively to determine how natural forces shape spatial structure and to evaluate whether the structure is holding its own against change. Trend information will be especially important in evaluating the speed and magnitude of changes in utilization and habitat quality in response to habitat improvements.

Although it seems clear that a good monitoring program for spatial structure will utilize both distribution and habitat information, relying too much on one type can be misleading. There are dangers in relying too heavily on just distribution and abundance or on just habitat quality and quantity. As already mentioned, the complexity of metapopulation dynamics can make distribution misleading. Heavily occupied areas that are actually sinks may be incorrectly regarded as highly productive patches; conversely, some vacant areas may actually be high-quality source patches that are not occupied because of the population's dispersal dynamics. Similarly, our knowledge of fish production-habitat relationships is too sparse to rely solely on a habitat approach. Especially problematic is a situation in which fish may be abundant but the habitat imposes sublethal effects that may impair the population's productivity. For example, low

levels of particular classes of pesticides may impair salmon's olfactory sensitivity, potentially interfering with homing and antipredator behavior (Scholz et al. 2000).

Although simple in concept, monitoring habitat aspects of spatial structure seems extremely open-ended in terms of what variables should be monitored, with what precision, at how many places, and how often. These are difficult questions, which need to be addressed in a specific monitoring document, but some basic guidance is appropriate here. The simple answer is that monitoring has to be done in such a way to make recovery biologists confident that the criteria they set are met. What variables should be monitored will be determined largely by the habitat challenges in a given basin. Temperature will probably be a major concern everywhere. But perhaps sedimentation will be important in some areas and not others. The same goes for flow, turbidity, and a host of other variables. Precision requirements depend, of course, on how small an effect needs to be detectable, and in some cases detectability may determine which variables are chosen. We expect that habitat quality and fish distribution monitoring will be done in index areas. Perhaps several panels of such areas can be set up in a basin, with one panel done each year. A critical question, of course, is whether this type monitoring should be done for every population. It certainly should for every population that has to meet abundance criteria, but not necessarily for others. It is probably not necessary to monitor every year. However, it seems logical that monitoring be done most intensively at the outset of the recovery effort, then less frequently as we gain understanding of the dynamics of the spatial structure. Monitoring aimed at spatial structure is obviously closely related to monitoring for juvenile outmigrants and for diversity, and taking advantage of these commonalities can decrease costs.

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APPENDIX A

ESU STRATA— ECOLOGICAL ZONE AND LIFE-HISTORY TYPE

The high degree of local adaptation exhibited by Pacific salmon is thought to be the product of their homing fidelity and the ecological diversity found in the Pacific Northwest. The life-history strategies exhibited by salmonids may reflect adaptations to specific freshwater environments. In the Willamette-Lower Columbia (WLC) recovery domain, there are (Table A.1):

- three major life-history types that are generally recognized for chinook salmon—spring run, fall run, and late-fall run;
- two types for steelhead—winter run and summer run; and
- one for chum salmon—fall run.

The WLC Technical Recovery Team (TRT) felt that each of these life-history strategies

Table A.1 Estimated number of populations in different strata in the WLC domain.

ESU	Ecological Zone ^a	Run Timing ^a	Historical Populations ^b
Lower Columbia chinook salmon	Coast Range	Fall	7
		Fall	9
		Late fall	2
		Spring	7
	Columbia Gorge	Fall	4
		Spring	2
Lower Columbia steelhead	Cascade	Summer	4
		Winter	14
	Columbia Gorge	Summer	2
		Winter	3
Columbia chum salmon	Coast Range	Fall	7
	Cascade	Fall	7
	Columbia Gorge	Fall	2
Upper Willamette chinook salmon	Willamette	Spring	7
Upper Willamette steelhead	Willamette	Winter	4
Total			82

^a Each ecological zone and run timing combination is a separate stratum.

^b The historical number of populations is based on Myers et al. (2002).

represented a substantial portion of the evolutionary legacy of the evolutionarily significant unit (ESU). Furthermore, there was concern that this subdivision of the ESUs still did not capture the essential diversity elements. For example, fall-run chinook salmon in the short, rainfall-influenced coastal tributaries were distinct from those in the large, snowmelt-influenced tributaries along the Cascade Crest, or the shorter tributaries in the rain shadow of the Cascade Crest. The concept of life-history/ecological strata was developed to describe this important level of between-population life-history diversity.

There are a number of methods of classifying freshwater, terrestrial, and climatic regions. Physiogeographic provinces were described from Washington and Oregon by Baldwin (1964), Fenneman (1931), and Easterbrook and Rahm (1970). Franklin and Dyrness (1973) identified natural vegetation zones in Oregon and Washington. The U.S. Environmental Protection Agency (EPA) has established a system of ecoregion designations based on soil content, topography, climate, potential vegetation, and land use (Omernik 1987). These ecoregions are similar to the physiographic provinces identified by the Pacific Northwest River Basins Commission (PNRBC 1969) for the Pacific Northwest. Furthermore, the National Marine Fisheries Service (NMFS) coastwide status reviews found a high correspondence between life-history trait distribution and Level III EPA ecoregions for coho salmon (Weitkamp et al. 1995), steelhead (Busby et al. 1996), chum salmon (Johnson et al. 1997), and chinook salmon (Myers et al. 1998). Additionally, Waples et al. 2001 indicated a high degree of correlation among ecology (as defined by modified ecoregions), life history, and biochemical genetics for Pacific salmonids. The TRT concluded that the EPA Level III ecoregions provided a useful measure of appropriate scale to characterize ecological diversity within the ESUs. However, the EPA ecoregions do correspond exactly to how salmon experience the environment. For example, all chinook in the Upper Willamette ESU use both the Willamette Valley and the Cascade EPA ecoregions, and it seemed reasonable, from a fish perspective, to identify a new ecological unit encompassing the eastern Willamette Basin above Willamette Falls. To avoid confusion with the EPA ecoregions, the TRT initiated the term ecological zone to describe the modified EPA ecoregions. Using the ecological zone as a reference, in combination with an understanding of the ecological features relevant to salmon, the WLC-TRT designated four ecological areas in the domain: (1) Coast Range zone, (2) Cascade zone, (3) Columbia Gorge zone, and (4) Willamette zone. The boundaries of these regions are shown in Figure A.1.

Another advantage of the ecological zone concept is that it provides geographic structure to the ESU. Maintaining each life-history type across the ecological zones reduces the probability of shared catastrophic risks. Additionally, ecological differences among zones reduce the impact of climate events across the entire ESU. The inclusion of a biologically based ESU substructure into possible recovery scenarios buffers the ESU against uniform declines.

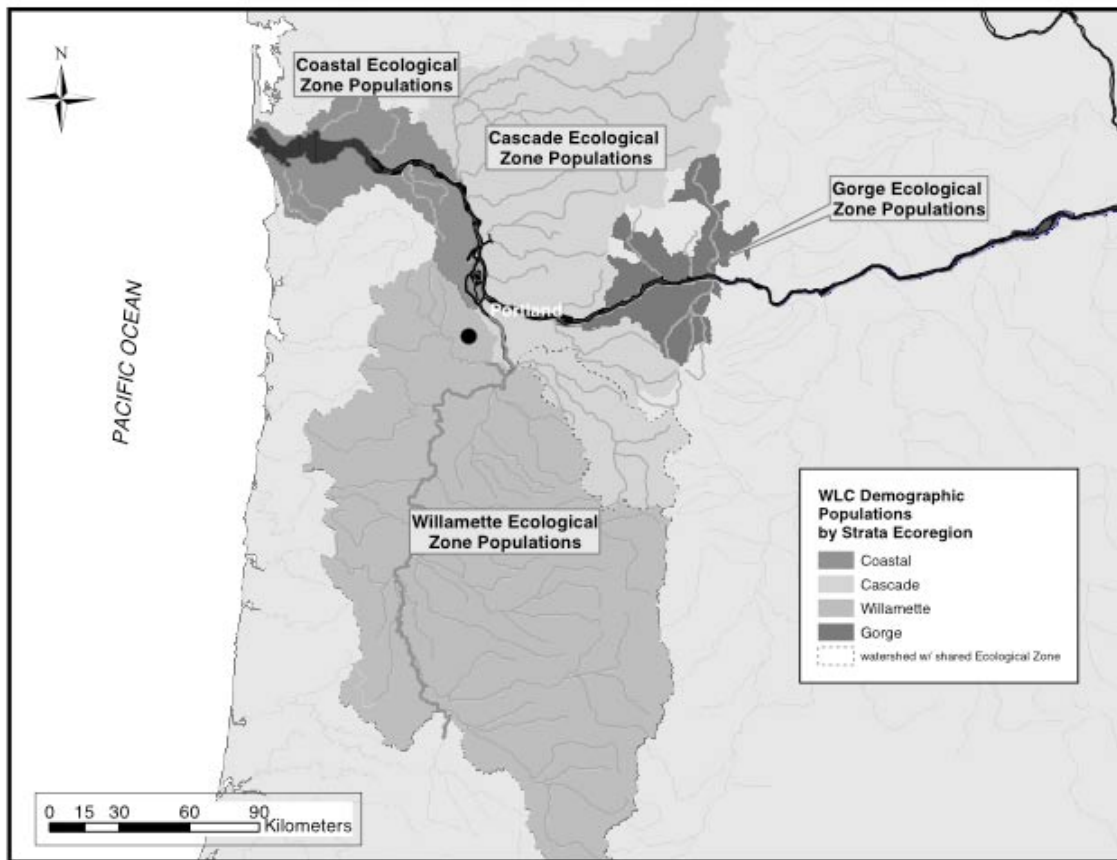


Figure A.1 Ecological zones within the WLC recovery domain. Ecological zones are based on EPA Level III ecoregions and NMFS ESU boundaries. Note that the Clackamas River Basin is shared by the Willamette and Cascade ecological zones.

Ecological Zones

The **Coast Range ecological zone** commences at the mouth of the Columbia River and extends upstream to the mouth of the Cowlitz River in Washington and to the mouth of the Willamette in Oregon (Figure A.1). In Washington these basins drain the Willapa Hills. The climate is dominated by moist Pacific marine air, with wet and mild winters and cool dry summers. Average annual precipitation is 200 to 240 cm, with approximately 80% occurring as rain between October and March. Streamflow is dependent on rainfall since drainage elevations do not exceed 500 m. River flows peak in December and January as a result of winter rainstorms. There is very little precipitation in July or August and corresponding flows are at the lowest with the highest water temperatures occurring in August. These rivers are especially prone to low flow during periods of drought. This region was originally forested with Sitka spruce, western hemlock, Douglas fir, and western red cedar.

The **Cascade ecological zone** extends from the mouth of the Cowlitz and Willamette Rivers to the Washougal and Sandy Rivers. Basin topography is dominated by the volcanic peaks of Mount Rainer, Mount St. Helens, Mount Adams, and Mount Hood, with each peak above 3,000 m. The lower portions of these basins pass through the Puget Lowland or Willamette ecoregion. At lower elevations, climate is similar to the Coast Range ecoregion, with wet mild winters and cool dry summers. Precipitation varies from 114 to 381 cm annually and generally occurs between October and March. Much of the precipitation above 1,000 m occurs as snow. In higher elevation basins, snowmelt adds to the surface runoff and provides a secondary peak in flow during the spring. Vegetation is similar to the Coast Range, except at higher elevations, where Pacific silver fir, noble fir, subalpine fir, mountain hemlock, and lodgepole pine occur.

The **Columbia Gorge ecological zone** encompasses the Columbia River Gorge, which extends from the mouth of the Sandy and Washougal Rivers to the Hood and White Salmon Rivers. The drainages in this region consist of short, steep streams that often limit anadromous passage to less than 1.6 km in smaller creeks. In larger systems, falls in the lower rivers often limit passage of salmon and steelhead. The climate and vegetation in this area are transitional between the high rain/snowfall area of the Cascades and the drier Columbia Plateau to the east. Stream flow is low in the summer except for larger basins influenced by snowmelt. Rivers in the Cascade and Gorge ecoregions are subject to catastrophic risks due to volcanic eruptions, such as occurred at Mount St. Helens in 1980.

The basins in the **Willamette ecological zone** occur in the Willamette Valley and Cascade ecoregions. In general this is an extension of the Cascade zone (Lower Columbia River ESUs), although it also includes presumptive populations that may have existed on the western slope of the Coast Range. The higher elevation portions of the Clackamas, Santiam, McKenzie, and Upper Willamette Rivers drain the Cascade ecoregion, which is described above. The remainder of the streams in the Willamette Valley occur in the Willamette Valley ecoregion. This ecoregion sits in a rain shadow, and annual precipitation is less than 120 cm. River flows peak in December and January, and low flows occur in August and September. Willamette Falls, at RM 42, was a natural barrier at low flow and only allowed for fish passage in the winter and spring for spring chinook and winter steelhead.

Life-History Types

Spring-run chinook salmon enter the Columbia in March and April, well in advance of their spawning time in August or September. Freshwater entry coincided with higher-than-average discharge in snowmelt rivers. Downstream migrant sampling suggests these fish migrate as subyearlings but enter the Columbia River later in the year than fall chinook.

Fall chinook salmon are divided into tule and bright populations. **Tule fall-run chinook salmon** spawn in all Lower Columbia River tributaries. Adults enter freshwater from August to October, with peak spawning in October. When tule fall Chinook enter freshwater, they are in their spawning colors.

Bright fall-run chinook salmon are present in the Lewis and Sandy Rivers. These fall chinook enter the Columbia River from September through January, with peak entry in October. Spawning time is protracted and fish have been observed spawning through the winter in the Lewis River. Current distribution of these fish is limited to the Lewis and Sandy basins.

Winter steelhead are present in most Lower Columbia River tributaries. These fish enter the Columbia River from November to May as mature fish. Spawning occurs from February through June with peak spawning in late April or early May.

Summer steelhead are present in the Kalama, Lewis, Washougal, Wind, and Hood Rivers. They enter freshwater as immature adults between March and October. All native summer steelhead in these basins historically occupied habitat above barrier falls, which excluded other salmon species.

Chum salmon spawn in most Columbia River tributaries from the mouth to Celilo Falls. Presently, chum salmon return to freshwater from October through December. Historic catch data indicate that chum salmon were occasionally caught in August. These may have been the early portion of the large fall run, or they may have been summer chum salmon.

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APPENDIX B

SUMMARY OF CORE AND GENETIC LEGACY POPULATIONS

Historically, each evolutionarily significant unit (ESU) was characterized by a number of populations that represented the substantial portion of the ESU's abundance or contained life-history strategies that were specific to the ESU. These core populations are important components to maintaining the evolutionary legacy of the ESU. The Willamette Lower Columbia Technical Recovery Team (WLC-TRT) concluded that recovery agencies consider giving priority to these core populations in developing their recovery plans. In addition to sustaining the evolutionary legacy of the ESU, these core populations may offer the most likely path to recovery. If these populations sustained large populations historically, they may have the intrinsic capacity to sustain large populations into the future.

Populations are considered genetic legacies for two reasons. The population may have had minimal influence from nonendemic fish due to artificial propagation activities, or the population may exhibit important life-history characteristics that are no longer found throughout much of their historical range in the ESU. Populations that are determined to be genetic legacies should be considered for prioritization in recovery efforts because they retain the most intact representatives of the genetic character of the ESU. Furthermore, populations that have maintained their genetic integrity should have retained a high degree of adaptation to local watershed conditions and are therefore more likely to achieve viable salmonid population (VSP) sustainability than are newly introduced or domesticated populations.

Table B.1 Historical Lower Columbia River fall run chinook salmon populations.

Population	Core Population (C)	Genetic Legacy (G)
Coast Range		
Youngs Bay		
Grays River		
Big Creek	C	
Elochoman River	C	
Clatskanie River		
Mill Creek		
Scappoose Creek		
Cascade		
Upper Cowlitz River		
Lower Cowlitz River	C	
Coweeman River		G
Toutle River	C	
Kalama River		
Salmon Creek /Lewis River		G
Lewis River late	C	G
Clackamas River	C	
Washougal River		
Sandy River early		
Sandy River late	C	G
Columbia Gorge		
Lower gorge tributaries		
Upper gorge tributaries	C	
Big White Salmon River	C	
Hood River		

Table B.2 Historical Lower Columbia River spring-run chinook salmon populations.

Population	Core Population (C)	Genetic Legacy (G)
Cascade		
Upper Cowlitz River	C	G ^a
Cispus River	C	
Tilton River ^b		
Toutle River ^b		
Kalama River ^b		
Lewis River	C	
Sandy River	C	G
Columbia Gorge		
Big White Salmon	C	
Hood River		

^a Cowlitz Salmon Hatchery broodstock

^b These populations were probably historically sustainable, but there is still some debate concerning their status as demographically independent populations (DIPs).

Table B.3 Historical Upper Willamette River spring-run chinook salmon populations.

Population	Core Population (C)	Genetic Legacy (G)
Clackamas River	C	
Molalla River		
North Santiam River	C	
South Santiam River		
Calapooia River		
McKenzie River	C	G
Middle Fork Willamette River	C	

Table B.4 Historical Lower Columbia River winter steelhead populations.

Population	Core Population (C)	Genetic Legacy (G)
Cascade		
Cispus River	C	
Tilton River		
Upper Cowlitz River	C	G ^a
Lower Cowlitz River		
North Fork Toutle River (Green River)	C	
South Fork Toutle River		
Coweeman River		
Kalama River		
North Fork Lewis River	C	
East Fork Lewis River		
Clackamas River	C	
Salmon Creek		
Sandy River	C	
Washougal River		
Gorge		
Lower gorge tributaries		
Upper gorge tributaries		
Hood River	C	G

^a Cowlitz Salmon Hatchery late-run winter steelhead.

Table B.5 Historical provisional Lower Columbia River summer-run steelhead populations.

Population	Core Population (C)	Genetic Legacy (G)
Cascade		
Kalama River	C	
North Fork Lewis River		
East Fork Lewis River		G
Washougal River	C	G
Columbia Gorge		
Wind River	C	
Hood River		

Table B.6 Historical provisional Upper Willamette River winter-run steelhead populations.

Population	Core Population (C)	Genetic Legacy (G)
<i>Westside tributaries^a</i>		
Molalla River		
North Santiam River	C	G
South Santiam River	C	G
Calapooia River		

^a There is still some debate concerning the historical status of this population.

Table B.7 Historical Lower Columbia River chum salmon.

Population	Core Population (C)	Genetic Legacy (G)
Coastal		
Youngs Bay	C	
Grays River	C	G
Big Creek	C	
Elochoman River	C	
Clatskanie River		
Mill Creek		
Scappoose Creek		
Cascade		
Cowlitz River fall- run/summer run	C	G ?
Kalama River		
Salmon Creek		
Lewis River	C	
Clackamas River	C	
Washougal River		
Sandy River		
Columbia Gorge		
Lower gorge tributary	C	G
Upper gorge tributary		

APPENDIX C

METHOD FOR INTEGRATING ATTRIBUTES AND ASSESSING POPULATION RISK OF EXTINCTION

Evaluation of Population Attributes

The proposed approach for integrating population attributes involves first evaluating the status of each population attribute separately on a 0–4 scale, then integrating the individual attribute values into an overall assessment of population status. The population attribute scores are based on the persistence category descriptions provided in each attribute section of this document, somewhat similar to those found in Table C.1. For example, the population spatial structure would be evaluated based on whether it is consistent with a persistence probability that is high, low, or somewhere in between and assigned a 0–4 score accordingly. For some criteria (e.g., adult productivity and abundance and juvenile outmigrant [JOM] growth rate), it may be possible to provide more quantitative thresholds associated with each level on the 0–4 scale. For other attributes (e.g., within-population diversity), it may not be possible to identify *a priori* quantitative thresholds, and more reliance on professional judgment will be required to determine the appropriate category. Issues related to the characterization of individual attributes are discussed in the chapter on each attribute; however, the determination of each attribute persistence level will follow a standardized procedure.

The TRT considered a number of possible procedures. Ideally, attribute persistence levels could be determined in a highly quantitative manner; however, in almost all cases the quantity and quality of available information necessary to derive such formulae were lacking (and will continue to be deficient under existing monitoring programs). Furthermore, the biological relationships among population characteristics are poorly understood. Data quality was a major concern for the TRT, and we generally agreed that any population attribute measure needed to include some accounting for uncertainty due to poor data quality, in contrast to uncertainty due to environmental stochasticity. Furthermore, adjustments for poor data quality needed to be precautionary in nature and should be distinct from evaluations of the biological parameters.

Given the current limitations of available information and our present understanding of ecological and population factors and interactions, the TRT agreed that a panel of experts, using

Table C.1 Examples of population attribute level characteristics under different scenarios of information quantity and quality.

Score – Expert A	0	1	2	3	4
Scenario 1 – Low uncertainty	0	0	10	0	0
Scenario 2 – High Uncertainty	0	2	6	2	0
Scenario 3 – Low Uncertainty	0	0	6	4	0
Scenario 4 – High Uncertainty	0	2	4	2	2

the persistence criteria provided in this document for each attribute, would provide the most efficient method of assessing the status of populations. The panel would be composed of scientists involved in a diverse array of fields related to salmon biology and ecology.¹ It is possible, and desirable, that a more quantitative model be derived as more information is collected (although that is unlikely anytime in the near future).

Panel members would review existing documents and information related to the specific attribute being evaluated. This information could be in written form or as part of a series of presentations by resource co-managers or the TRT. Data interpretation and a review of overall data quality would be discussed by the panel prior to persistence level characterization. The method used to capture the view of each panel member regarding each attribute would be similar to the method used by the NOAA Fisheries Biological Review Team to make initial listing evaluations and based on an approach developed by FEMAT.² Each panel member would have ten votes to allocate into the five persistence levels (0–4) for that attribute according to the criteria and evaluation guidelines provided in each attribute section. The distribution of an individual's votes would reflect uncertainty regarding that level determination (e.g., Table C.1). In Table C.1 (scenario 1), the panel member concluded that the information available indicated that for population A the productivity and abundance persistence level should be a 2. By placing all 10 votes in the 2 box, the panel member was indicating a high degree of certainty in the score. In scenario 2 the information was less compelling, and the vote distribution was more broadly set around the mean of 2. Scenario 3 reflects a situation in which the population attribute status is intermediate between 2 and 3; in this case, the mean is 2.4. Panel members will use professional judgment to weight the factors used to arrive at the vote distribution. Finally, scenario 4 illustrates a situation in which the mean vote is still 2.3, but with greater uncertainty in the data.

Persistence levels for the population attribute would be calculated from the combined votes from all panel members (Table C.2). The attribute mean and vote distribution would be presented in describing the population attribute status. When expert panels are employed, voting

Table C.2 Hypothetical scoring of a population attribute and data quality by an expert panel.

Score						Data
Expert	0	1	2	3	4	Quality
A	0	7	3	0	0	1
B	0	4	5	1	0	2
C	0	5	5	0	0	2
D	0	4	5	1	0	3
E	0	6	4	0	0	1
F	0	5	5	0	0	2
G	0	8	2	0	0	1
H	1	7	2	1	0	1
Sums	1	45	31	3	0	1.6
Average	1.46	Poor data	(-0.25)	= 1.21		

¹ A similar body was assembled to evaluate the status of salmon ESUs as part of the listing process.

tables are useful in presenting the uncertainty underlying the evaluations. Additionally, panel members would consider the quality of data utilized to determine the attribute status. Data quality would be scored from 0 to 4, 4 being high-quality data with little measure error. If the panel determined that the data quality was especially poor (0, 1, or 2), they could decide to reduce the population attribute mean as a precautionary measure. The amount of the reduction would be directly related to the data quality score. In the extreme case, where no information exists on a population attribute, the panel may use correlated information to arrive at a score. For example, in a population where the adult productivity and abundance attribute is categorized as 2.8, one might infer that the JOM attribute level would be similar. Where only correlated information is available, the data quality measure would be characterized as being very low. This would lead to a severe reduction in the mean persistence levels for attributes that have not been directly monitored. Further guidelines for calculating persistence levels for attributes with no data are provided in the “Combining Population Attributes” section of this appendix.

Approach to Integrating Population Attributes

Each attribute contributes to a population’s viability assessment. Integrating the attributes into a single population persistence level needs to be done in a manner that weighs the relative importance of each attribute. The TRT, in general, concluded that the productivity and abundance metric provided the most direct and objective measure of population viability. The productivity and abundance persistence level was weighted twice as heavily as the other attributes. Additionally, the attributes were grouped into two category types: attributes that describe the population’s performance (productivity and abundance and JOM) or the population’s potential (population diversity, spatial structure, and habitat). Both performance and potential are essential to a population’s viability. Attributes within a category are thought to be highly correlated, and in those cases where no data are available for an attribute, the other attribute(s) in the category provide the most appropriate source of information. Where no attributes exist in a category, neither the category nor the population can be evaluated. Additionally, if any attribute level is categorized as a 0 (in contrast to no data), then the population persistence category must be 0. For example, if there is good habitat, but monitoring indicates that there are no fish present (i.e., in the presence of an impassible barrier), the population has no possibility of persistence. Alternatively, the presence of attributes with means of 0 may indicate problems in data interpretation. If adults are monitored on the spawning grounds, but no juveniles are observed emigrating, the adults may represent strays from other populations or juvenile monitoring is not effectively capturing outmigrants. Under most conditions, however, the population persistence level would be computed according to the formula:

$$\text{Population} = (\text{Performance Attributes}) + (\text{Potential (sustainability) Attributes})$$

as calculated from:

² Forest Ecosystem Management Assessment Team (<http://www.environment.pdx.edu/fem.htm>).

$$Population = \left(\frac{1}{3}(G \& A) + \frac{1}{6}(JOM) \right) + \left(\frac{1}{6}(Space) + \frac{1}{6}(Diveristy) + \frac{1}{6}(Habitat) \right)$$

In addition to the computation of a single population persistence level, it is more informative to present the persistence levels of the component attributes. This can be done in either graphic form (Figure C.1) or tabular form (Table C.3). For those attribute levels that were derived using poor quality data, the magnitude of any reduction is clearly indicated (as an incentive for improved monitoring). This method confers most of the information used to derive the population persistence levels. It is intended that recovery entities would utilize this information to prioritize actions that would bring the population persistence level to VSP status. Population persistence levels would be used to estimate strata persistence levels and overall ESU viability as described in previous sections.

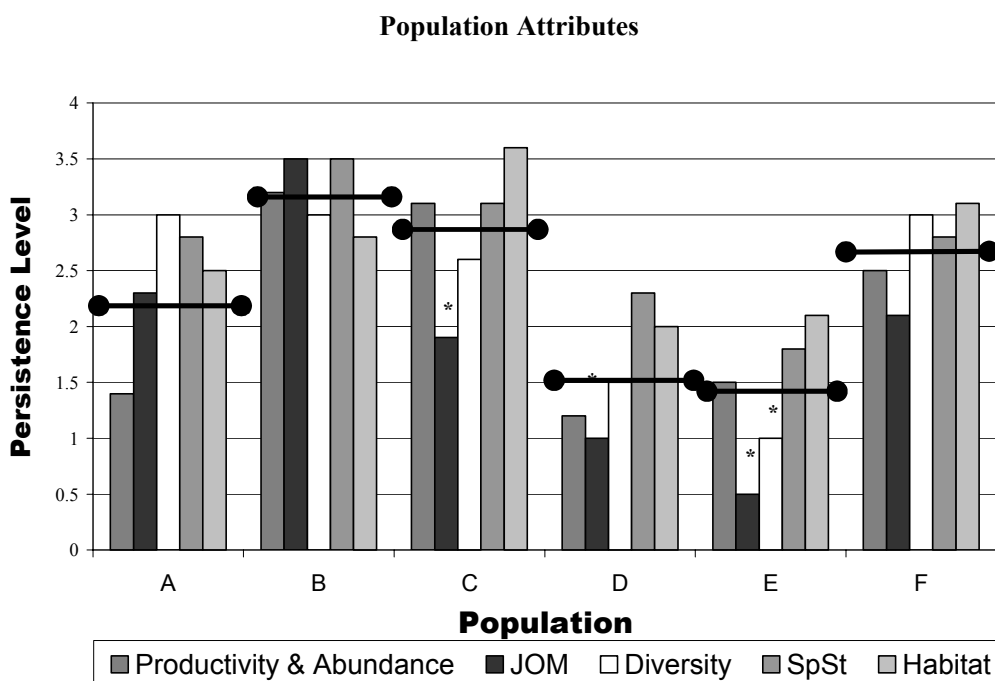


Figure C.1 Example of a graphical display of population attribute persistence levels for populations A–F. The values used are fictional, and not meant to represent any strata or ESU. The dashed lines indicate the overall population persistence levels. Population means were derived using the weighed average algorithm presented above. Asterisks indicate the attribute mean prior to any reduction for poor data quality.

Table C.3 Tabular representation of the information presented in Figure C.1. Population persistence levels are derived from the attribute means using the weighted average algorithm presented earlier in this section. Bracketed numbers following the attribute mean indicates the reduction in attribute mean due to poor data quality.

Population	Population Attribute Persistence Categories					Population Persistence Category
	Productivity and Abundance	JOM Growth	Spatial Structure	Diversity	Habitat	
A	1.4	2.3	3.0	2.8	2.5	2.22
B	3.2	3.5	3.0	3.5	2.8	3.18
C	3.1	1.9 (0.25)	2.6	3.1	3.6	2.88
D	1.2	1.0 (0.50)	1.5	2.3	2.0	1.52
E	1.5	0.5 (0.40)	1.0 (0.30)	1.8	2.1	1.39
F	2.5	2.1	3	2.8	3.1	2.65
Strata						=1.98

APPENDIX D

POPULATION CHANGE CRITERIA

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Population Change Criteria Overview

The population change criteria (PCC) provide a novel performance test for evaluating whether a threatened population has recovered and is no longer in danger of extinction. The approach starts with the development of a viability curve, which describes the relationships among population abundance, productivity, and extinction risk (Figure D.1). The extinction risk experienced by a population is a function of both the population's productivity and size (Musick 1999, McElhany et al. 2000). We define productivity as the number of returns produced per spawner, when the population is at low density relative to carrying capacity. All else being equal, a population with a high average productivity could persist at a lower abundance than a population with a low average productivity. This is because a population with high average productivity would have a higher probability of returning to the original abundance if perturbed to low abundance than a population with low average productivity. A high-productivity population could be characterized as being more resilient than a low-productivity population. The amount of environmental variation affects the likelihood that a population will be perturbed to low abundance and is another key parameter in the estimation of extinction risk. With regard to population size, all else being equal, the smaller a population is, the more likely it is to fluctuate to extinction (Thomas 1990, Lande 1993). The viability curve can be estimated using a population projection model that incorporates abundance, productivity, environmental

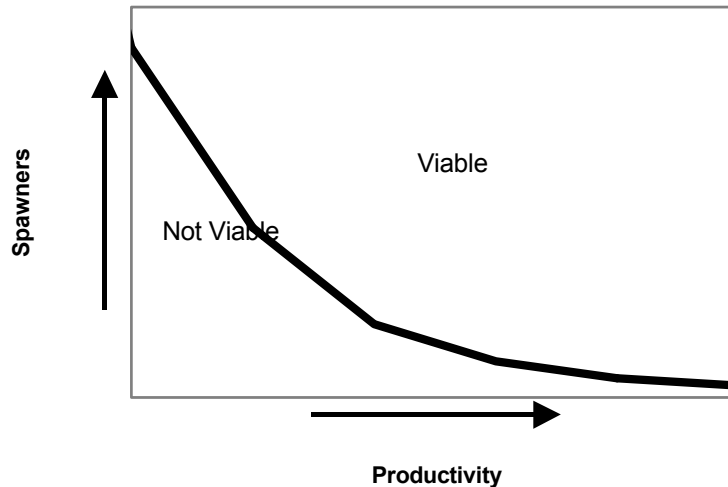


Figure D.1 The relationship between productivity, population size, and extinction risk. The curve represents combinations of size and productivity that have exactly the acceptable extinction risk.

variability, and any other factor considered relevant for estimating extinction risk.

Key issues with developing criteria from viability curves are determining an appropriate form for the population projection model and determining how to estimate parameters. As described in the first section of this appendix, the projection model used for the PCC viability curve is relatively simple and is well described in the population dynamics literature. The next section of this appendix (“Projection Model and Minimum-Size Estimation Methods”) describes the distinguishing features of the PCC approach. These features involve the method used to estimate productivity and the development of a population performance test.

Projection Model and Minimum-Size Estimation Methods

Model Overview

We calculated a viability curve using a population projection model of stochastic exponential growth with a ceiling and a lower critical threshold (Figures D.2 through D.5).

$$\begin{aligned} N_{t+1} &= 0 \quad \text{if } N_t \leq QET \\ N_{t+1} &= N_t e^r \quad \text{if } QET < N_t \leq k \\ N_{t+1} &= ke^r \quad \text{if } N_t > k \\ \text{where } r &\approx \text{Normal}(\mu, \sigma). \end{aligned} \tag{Eq. 1}$$

where

N_t is the population size at time t ,

k is the maximum size of the reproductive population (i.e., “ceiling”),

r is a stochastic parameter describing the per capita reproductive rate, and

QET is the quasi-extinction threshold.

The parameter μ is the median per capita growth rate of a population below k , and σ^2 describes the environmental variability in growth rate (“process variance”). The normal distribution of r is a theoretical consequence of the central limit theorem applied to a multiplicative survival process (Hilborn and Walters 1992). In the nomenclature of recruitment models, this is a “stochastic hockey-stick” model, as compared to a Ricker or Beverton-Holt model (Barrowman and Myers 2000). The median annual growth rate, λ , for a population below k is $\lambda = e^\mu$. We will refer to the median growth rate of a population below k as the productivity of the population, and represent productivity with the symbol γ . The Ricker and Beverton-Holt recruitment models have a productivity parameter often symbolized as α , which represents the “intrinsic productivity” or number of returns per spawner if there was only one spawner (Hilborn and Walters 1992). Since the interpretation and values of the parameters in the hockey-stick and the other models differ, we have adopted a different symbol to avoid confusion. If $\gamma > 1$, the equilibrium mean abundance with this model is near k . If $\gamma < 1$, the equilibrium mean abundance is 0 (extinction). Extinction risk using the model is estimated as the probability that a population starting at some initial population size, N_0 , declines to the QET within a given time horizon. The extinction risk is estimated by simulating the population process with some given growth rate and process variance to produce many population trajectories, then calculating the fraction of simulated population trajectories that declined to QET within the specified time period.

Because of the age structure of salmon populations, the population dynamics model was applied to a four-year running sum of annual spawner counts as described in Holmes (2001) and McClure et al. (in review). Thus,

$$N_t = \sum_{i=0}^3 S_{t-i} ,$$

where N_t is as above and S_t is the number of spawners in year t . Both initial population sizes and QET are stipulated in the model in terms of four-year sums, which is equivalent to an average annual spawner count over four years of $N/4$.

Using this model, we identify the minimum population size for a given productivity as the initial population size, N_0 , which just produces an acceptable extinction risk (Figure D.2). The minimum size is found using a simple search algorithm that tests the extinction risk associated with a number of different potential initial population sizes. If a population were to start out at a size smaller than the minimum size, the extinction risk would be too high; and if the initial population size were larger, the extinction risk would be lower than the acceptable risk originally specified. The variance parameter of the model, σ^2 , is an empirical estimate based on recent historical abundance time-series data for the population or species (see below for estimation approach). The population ceiling, k , is set as the initial population size. Thus, we estimate the minimum population size under the scenario that the minimum population size is also the population ceiling. This effectively allows the minimum population size estimate to also be an estimate of minimum carrying capacity. We can seldom estimate with confidence the carrying capacity of a population, and this approach provides a precautionary estimate of the minimum population size, since a population constrained by a low ceiling has a higher extinction risk than a population without a ceiling.

This is a very simplified model of salmonid dynamics, which does not include many of the features associated with salmon biology, such as ocean regime shifts, short-term temporal autocorrelations, complex recruitment functions, etc. We addressed these issues in a variety of ways, and the final criteria reflect consideration of more factors than are reflected in Equation 1 alone.

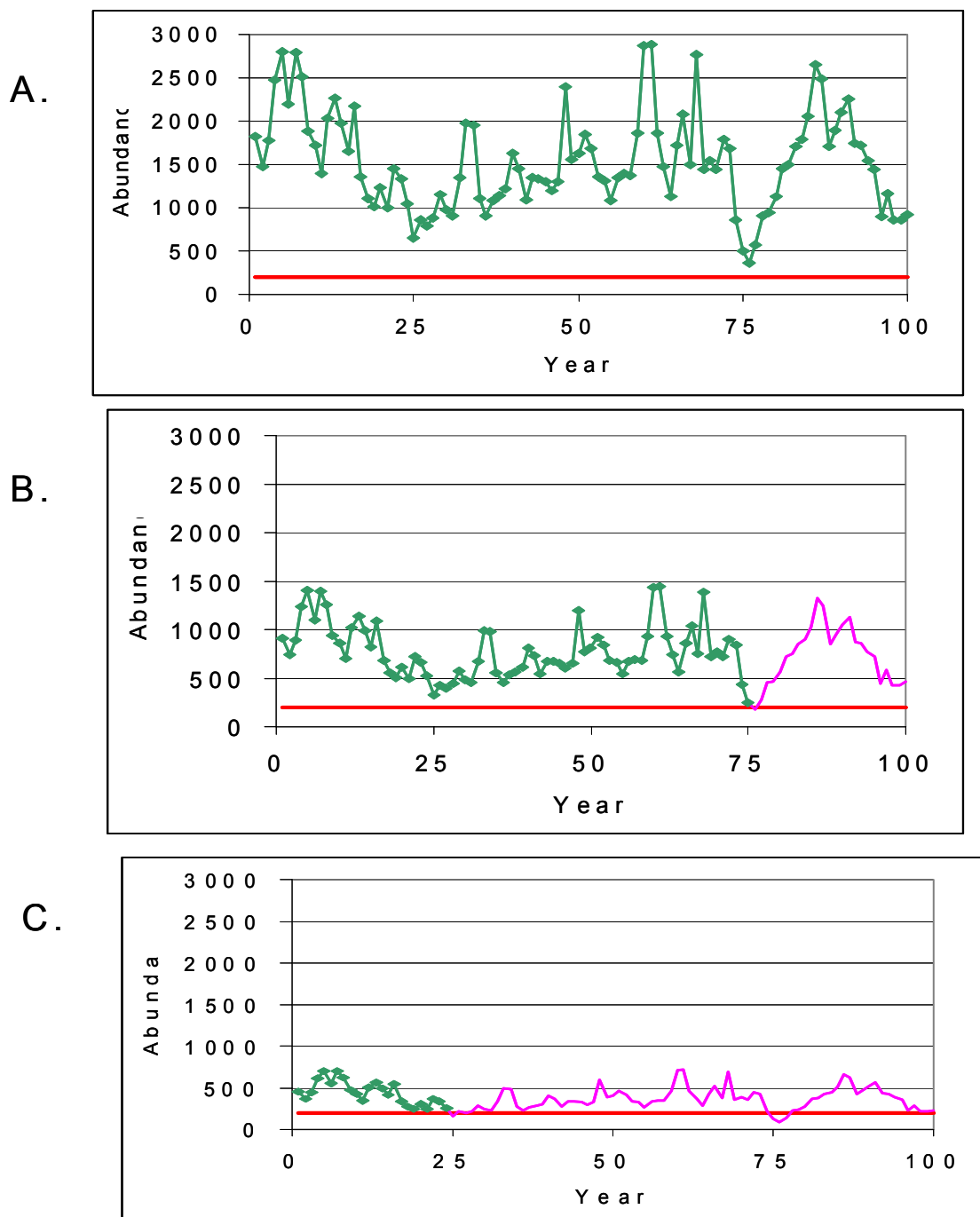


Figure D.2 Simulated population trajectories illustrating the relationship between population abundance, environmental variability, and extinction risk. The lower line indicates the quasi-extinction threshold (QET); populations that drop below this level are considered functionally extinct.

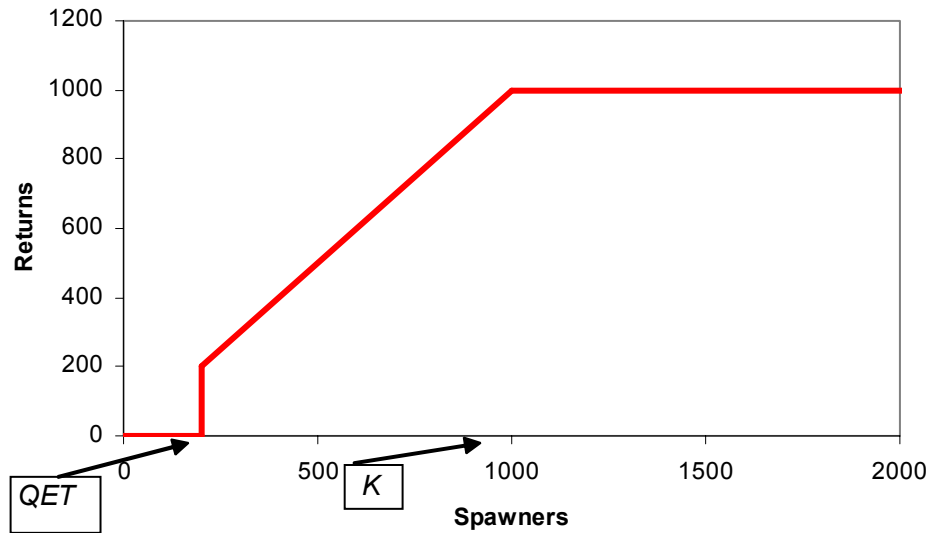


Figure D.3 Conceptual drawing of recruitment function for projection model to identify minimum population size. This is a hockey-stick model, with a depensitory threshold. Below QET spawners, the population is considered extinct. Above k spawners, the returns are constant. The slope of the line at abundances between QET and k is an indication of the productivity of the population (γ). This graph represents only the deterministic skeleton of the model. Productivity is actually a stochastic variable driven by environmental variation.

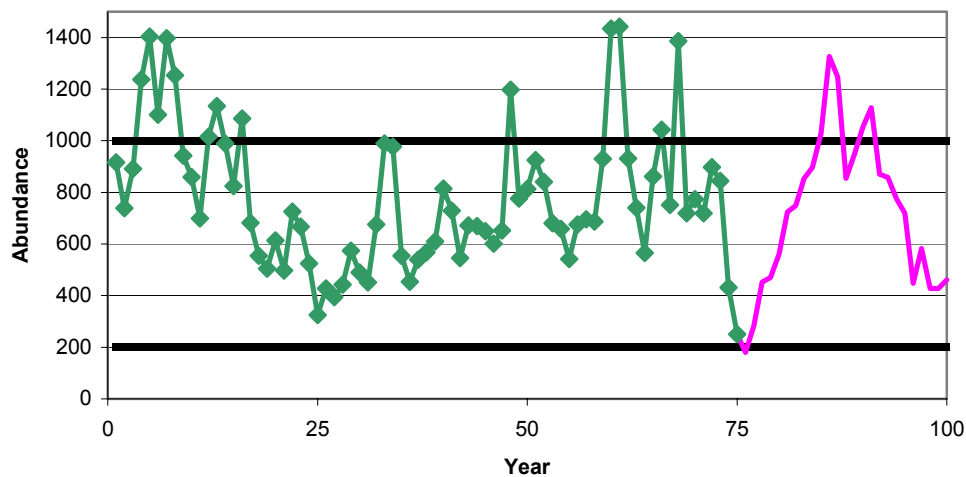


Figure D.4 Simulated trajectory showing the dynamics of the population dynamics model. The upper dashed line represents k and the lower dashed line represents QET . Once the population goes below QET , it is considered functionally extinct, but the trajectory in the diagram continues in order to show the future dynamics had a lower QET been selected. Because this is a stochastic model, it is possible for a population to temporarily exceed k , but k does constrain the upper size of the population.

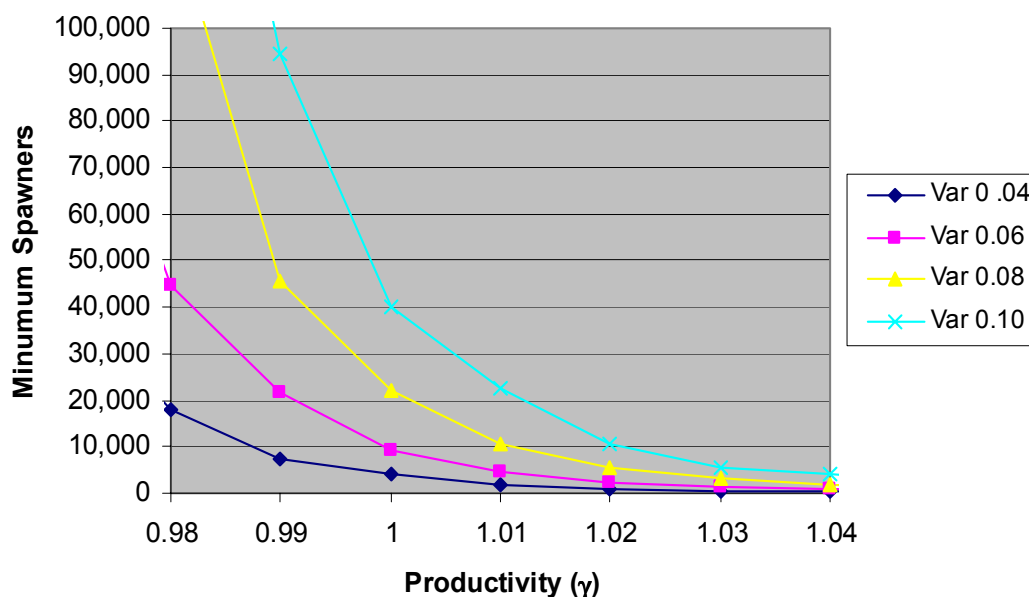


Figure D.5 Viability curves for populations with different values of environmental variability. The acceptable risk is a 5% probability of declining to a four-year average of 50 spawners in 100 years.

Specifying the Acceptable Risk

This criteria approach requires the specification of an acceptable extinction risk. The acceptable risk can be stated as the probability that the population will decline to QET individuals in “time horizon” years. The probability and time horizon parameters are largely policy decisions about acceptable risk, and options regarding these values are presented in this document. The QET should have some biological meaning. This is the population size below which depensatory (Allee) effects are believed to be so strong that extinction risk greatly increases because of processes in addition to environmental stochasticity, or that uncertainty about population behavior becomes unacceptably high (Dennis et al. 1991). This is an extremely difficult parameter to estimate, and the consequences of this parameter estimation problem are discussed below.

Setting QET

Some of the processes that may be important in setting the QET are inbreeding depression, loss of genetic diversity, ecological Allee effects, mate finding, and demographic stochasticity (Goodman 1987, Lande 1998). Of these processes, we set QET at an abundance that

avoids potential negative effects from demographic stochasticity and the loss of genetic diversity.

Demographic stochasticity refers to variability in fitness (family size) among individuals, whereas environmental stochasticity refers to environmental variability that affects the mean fitness of the entire population (Lande 1998). The individual variability only tends to affect extinction risk at very small population sizes, because at larger sizes individual variations average out and environmental stochasticity dominates. Demographic stochasticity can lead to increased extinction risk of small populations, because even if the environment is constant, chance variations in family size may result in reproductive failure of all individuals in a single year. Risk from demographic stochasticity is also influenced by chance variations in sex ratio (i.e., there is some probability that only one gender will return). To inform our choice of QET, we explored an individually based simulation model that identified an abundance above which a population is expected to be relatively immune from risks associated with demographic stochasticity caused by variations in family size and chance fluctuations in sex ratio (McElhany and Payne in prep). This model suggested that if a population stays above 40 spawners in a given year, it is likely to experience little additional extinction risk from demographic stochasticity over 100 years. This finding is similar to other studies of risks from demographic stochasticity (Lande 1998).

A number of theoretical and empirical studies relate extinction risk and loss of genetic diversity (e.g., Soule 1980, Thomas et al. 1996, Keller and Waller 2002). As one measure of genetic diversity, the rate of loss of neutral alleles can inform our selection of QET, though it is difficult to make direct links between the loss of neutral alleles and population viability. Published studies on the loss of genetic diversity in small population sizes suggest that at effective population sizes below about 50, there is a relatively high probability of the loss of neutral alleles due to genetic drift (Soule 1980). The effective population size, N_e , is a genetic term referring to number of individuals required if the population had an “ideal” mating system (Wright 1938). The effective size of a population is generally smaller than the census count of the population (Waples 1990a and 1990b) and by assuming an average generation time of five years and an effective population size to census count ratio of 0.2, the Puget Sound Technical Recovery Team developed a recommended QET of an average of 62.5 spawners per year for four years (PS-TRT 2002).

Both the demographic stochasticity and genetic loss approaches suggest that extinction risk is affected by deleterious processes in addition to environmental stochasticity at population sizes below about 50 spawners in a given year. Therefore, we used a QET value of 50 spawners per year for estimating growth rate and abundance viability criteria. This annual spawner count threshold translates to a QET of 200 in the four-year running-sum model (Eq. 1).

Estimating Variance

After the acceptable risk statement is specified, the only parameter used to derive the estimation of the minimum population size for a given productivity is the estimate of environmental variance. Environmental variance is the variance parameter describing the distribution of r in equation 1. If we assume that perfect abundance counts are available and that a population is not experiencing density dependence, the variance parameter can be estimated from an abundance time series as (Dennis et al. 1991):

$$\hat{\sigma}^2 = \text{var} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right). \quad \text{Eq. 2}$$

If the population is near some density-dependent carrying capacity, this equation will tend to underestimate the environmental variance parameter in equation 1. Because the recent historical time series used to estimate the environmental variance typically contain large measurement errors, we employed the slope method variance estimation technique developed by Holmes (2001). This method helps correct for the large upward bias in the variance estimate that is produced by measurement error. The slope method equation is:

$$\hat{\sigma}^2 = \text{slope of } \text{var} \left(\ln \left(\frac{N_{t+\tau}}{N_t} \right) \right) \text{ vs. } \tau, \quad \text{Eq. 3}$$

where τ is the temporal lag between the values used for the variance estimate. For our variance estimations, we estimated the slope based on a maximum τ of 4.

The variance estimate is just that, an estimate. Because we assume, based on theoretical and empirical considerations, that $\ln(N_{t+1}/N_t)$ is normally distributed, we have an estimate of the sampling distribution of $\hat{\sigma}^2$. The sampling distribution of the variance of a normally distributed random variable is:

$$\sigma^2 \approx \frac{\hat{\sigma}^2 * df}{X_{df}^2}, \quad \text{Eq. 4}$$

where df is the sample degrees of freedom, and X_{df}^2 is a chi square distribution with df degrees of freedom (Sokal and Rohlf 1981). If the variance is estimated using perfect abundance counts and equation 2, the degrees of freedom is equal to the number of N_{t+1}/N_t ratios minus 1. If four-year running sums are used, the degrees of freedom would be the number of annual spawner counts minus 4. Variance estimates calculated with the slope method have this same distributional form, but the degrees of freedom are reduced (Holmes and Fagan 2002). Although the slope method reduces bias in the variance estimate associated with measurement error, it does so at a cost of decreased precision. Holmes and Fagan (2002) have calculated tables for determining the degrees of freedom associated with slope method variance estimates.

It is likely that, because of unique circumstances, every population has a unique environmental variance value. However, the variance estimate for any particular population is often extremely uncertain because available time-series data sets are short relative to the levels of variability. If we assume that the populations within an evolutionarily significant unit (ESU) tend to experience similar levels of environmental variation, we can obtain a potentially more accurate and precise estimate of the variance by “pooling” variance estimates from multiple populations. If it is assumed that there is a single true environmental variance value that is common to every population in an ESU and that every population time series represents an independent sample of that variance, the average of all the population estimates provides an unbiased estimate of the true variance, and the sample distribution has the degrees of freedom equal to the sum of the degrees of freedom from each individual population estimate. Under the

assumption that all populations experience basically the same levels of environmental variation, the differences in observed variance estimates for individual populations represent a form of sampling error and do not necessarily reflect true differences in variation.

In calculating the minimum population size, we are interested in the natural levels of environmental variation that will be present no matter what hatchery or harvest management strategy is employed. Hatcheries and harvests have the potential to obscure estimates of natural environmental variation if we simply look at number of spawners on the spawning ground. Therefore, in our approach we have incorporated a way of partitioning out the variance changes induced by hatcheries and harvest (McClure et al, McElhany and Payne in prep). We single out hatcheries and harvest for this variance correction process partially because we can measure the effect, but primarily because we have an *a priori* expectation that hatcheries and harvest will alter the level of variation observed on the spawning ground since most harvest strategies explicitly or implicitly seek to reduce variation in escapement and hatcheries are likewise expected to affect observed levels of variance. These variance estimation details are presented in Appendix E.

The variance estimation approach assumes that the historical time series is not experiencing density dependence. If the historical time series represents a population at carrying capacity, then the variance estimate describes the variability in carrying capacity and survival. It is not clear whether this variance estimate would be higher or lower than the variance observed if a population were not experiencing density dependence. If the carrying capacity is fairly stable, the variance estimate calculated for a population near carrying capacity would tend to underestimate the variance of the population abundance below carrying capacity. The power to detect density dependence is generally pretty low (Dennis and Taper 1994, Appendix G this document), which increases our uncertainty about the variance estimate. Given that many populations are declining, it seems reasonable to assume that they are below capacity and are declining, because survivals are too low for replacement; however, the populations could simply be tracking a declining capacity.

Using recent time series to estimate levels of environmental variation for modeling future population dynamics carries the explicit assumption that the recent past will be predictive of future levels of environmental variation (stationarity assumption). Human actions can affect environmental variation, and the future may not resemble the past, but we cannot predict the magnitude or direction of potential change. In general, the viability criteria are determined assuming that the past is a good predictor of future behavior of salmon populations. To the extent that this assumption is violated, the criterion will need to be reevaluated. We obviously will not know the extent to which the assumption is violated until the future happens. It is important to actively test the model's assumptions.

PCC Targets

PCC Targets Overview

If the demographic model and viability curves are going to be employed to establish viability criteria, it is necessary to somehow estimate population productivity. The viability of a population is a function of both the population size and productivity. Therefore, both population size and productivity will need to be evaluated in the future to determine whether currently listed populations have achieved viable status.

The traditional fisheries approach to estimating productivity relies on fitting recent time-series data to stock-recruitment functions such as the Ricker, Beverton-Holt, or hockey-stick models (Hilborn and Walters 1992, Appendix G this document). However, there is generally very little statistical power to estimate productivity with the stock-recruitment model fitting approach (Hilborn and Walters 1992, Appendix G this document). In fact, it is often impossible to even determine whether or not a population has experienced density dependence near capacity over the observed time period (Dennis and Taper 1994, Hooten 1995, Ray and Hastings 1996, Shenk et al. 1998, McClure et al. in review). The conclusion researchers tend to reach regarding whether or not a population is at carrying capacity depends on prior assumptions and on how the question is asked. If the null hypothesis (prior assumption) is that the population *is not* experiencing density dependence, the hypothesis is generally very difficult to disprove. If the null hypothesis (prior assumption) is that the population *is* experiencing density dependence, that hypothesis is also generally very hard to disprove. Accurately and precisely estimating intrinsic productivity is even more challenging than testing hypotheses about carrying capacity because estimating intrinsic productivity requires extrapolation to predict recruitment at very low (i.e., < 1 fish) spawner abundances (Hilborn and Walters 1992). There is seldom much data at these low abundances to support the extrapolations. The extrapolations tend to depend critically on the exact form of the recruitment function employed, and there is often little statistical power to distinguish among different possible recruitment functions (Appendix G). An understanding of the limitations of recruitment curve fitting would be greatly advanced if confidence intervals or probability distributions were commonly reported for parameter estimates of intrinsic productivity, and if formal model selection methods (e.g., Akaike's Information Criterion (AIC)) were adopted. Although in some situations data clearly convey a particular stock-recruitment relationship, they tend to be the exception rather than the rule.

As an alternative to fitting stock-recruitment functions, we have relied on estimates of the population growth rate (observed λ) as a measure of population productivity (γ). The observed growth rate of a population is a precautionary estimate of population productivity, in that the productivity is unlikely to be lower than the observed growth rate, but it may very well be higher. If a population is below carrying capacity, it can grow as a result of increased survival, in which case λ is, by definition, an appropriate estimate of γ (Table D.1). If a population is near carrying capacity, population growth requires an increase in capacity. The γ value for a population tracking an increase in capacity may be expected to be at least equal to its observed growth rate, though it may be higher.

Table D.1 Possible relationships between median annual growth rate and intrinsic productivity.

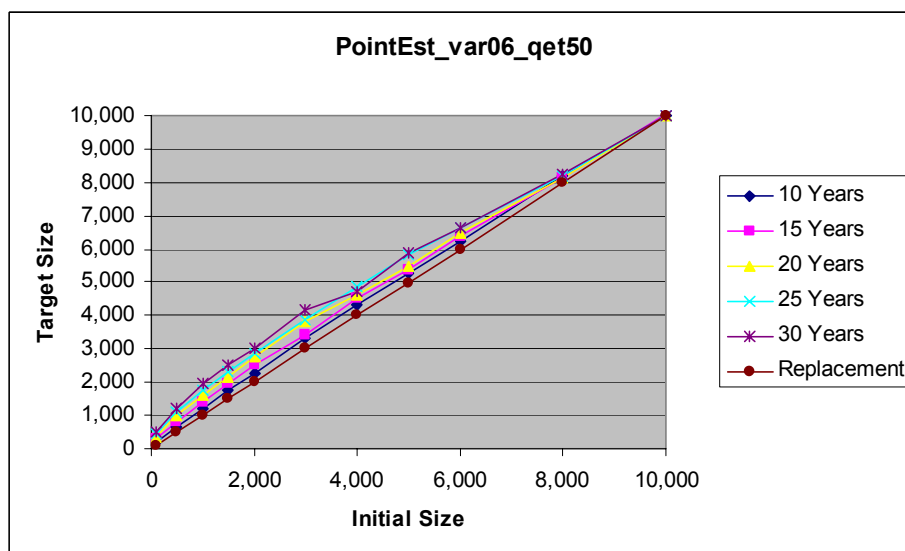
Observed Median Annual Growth Rate (λ)	Carrying Capacity (k)	Intrinsic Productivity (γ)	Interpretation
$\lambda < 1$	$N < k$	$\gamma = \lambda < 1$	Population below carrying capacity and declining because of low survival.
	$N = k$; k declining	$\gamma \geq 1$; γ may be $> \lambda$	Population tracking a declining carrying capacity.
$\lambda = 1$	$N < k$	$\gamma = \lambda = 1$	Population below carrying capacity and productivity just at replacement.
	$N = k$; k stable	$\gamma \geq 1$; γ may be $> \lambda$	Population has relatively high intrinsic productivity and is fluctuating around capacity.
$\lambda > 1$	$N < k$	$\gamma = \lambda > 1$	Population below capacity, improvement in survival produces productivity greater than 1. Population will stabilize ($\lambda = 1$) once it reaches capacity.
	$N = k$; k increasing	$\gamma > 1$;	Population has relatively high intrinsic productivity and is tracking an increasing capacity.

It is possible to calculate in advance the growth rate associated with a particular change in population size over a specified period of time using the equation

$$\hat{\lambda} = \exp \left(\frac{\ln \left(\frac{\phi}{\iota} \right)}{y} \right). \quad \text{Eq. 5,}$$

where ι is the initial population size, ϕ is the final population size, and y is the number of years between observations. For example, if a population increased from a four-year average annual abundance of 1,000 spawners to 1,800 in 20 years, the point estimate of λ ($= \gamma$) would be 1.033. In addition, the spawner abundance at the end of the 20 years would be 1,800. This ability to estimate productivity associated with a given increase in population size allows for the calculation of the PCC (Figures D.6 and D.7). With PCC, we ask, “Given the current population size, how big does the population need to be in Y years to have demonstrated a productivity and abundance that gives an acceptable risk?” This future population size that gives an acceptable risk we refer to as the target size for the population in Y years. The target size of a population is a function of the current size of the population, the environmental variance of the population, the acceptable risk statement, and the number of years in which to reach the target. The target size is found using a search algorithm that examines the extinction risk associated with a number of different potential target sizes before identifying the target size with the specified acceptable risk.

A.



B.

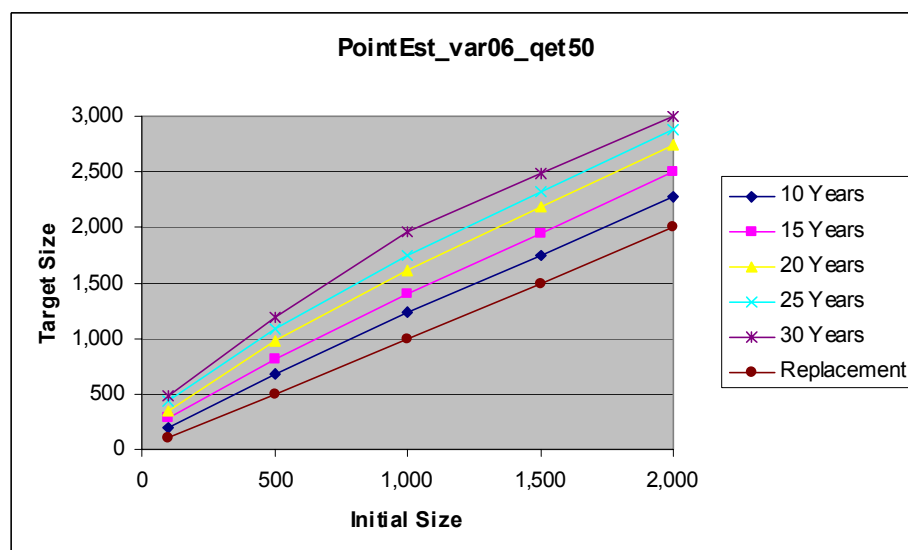
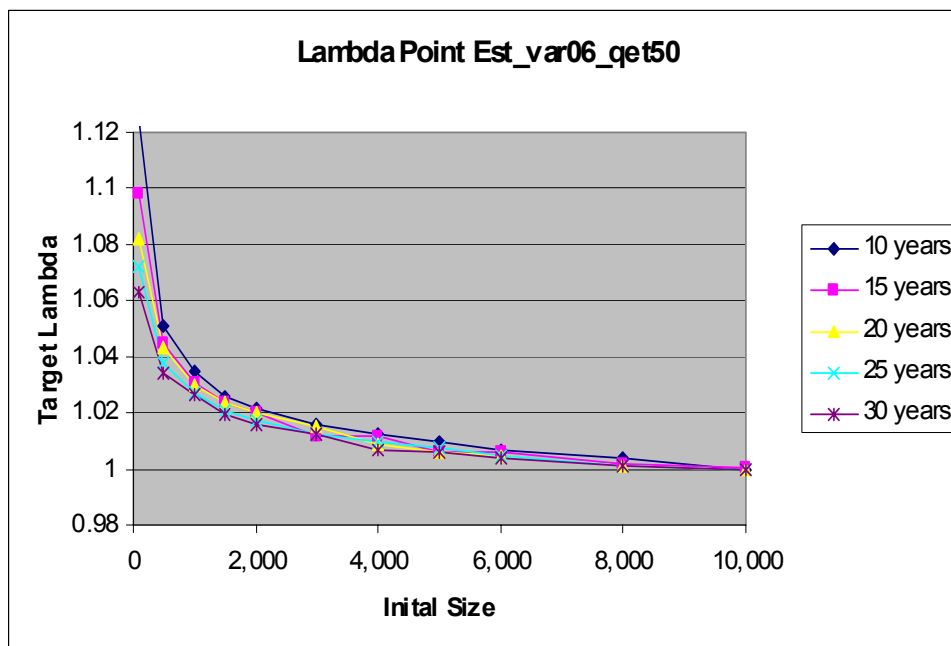


Figure D.6 Population growth criteria based on point estimates of λ and σ^2 . The σ^2 value was 0.06.

Panel B shows an expansion of the lower portion of the x axis of panel A. The target size is that which a population needs to achieve in a given time to have a productivity ($\gamma = \lambda$) that has an acceptable extinction risk. All curves in the diagram represent a 5% probability of declining to a four-year average of 50 spawners in 100 years. The years in the different curves are the number of years to reach the target size from the initial size. The “replacement” curve is for reference purposes; it indicates where the target size equals the initial size.

The PCC targets may be expressed equivalently as either a target abundance in a given number of years when starting from a given initial abundance (i.e., ϕ in Eq. 5) or as a population growth rate when starting from a given abundance (i.e., $\hat{\lambda}$ in Eq. 5). In this appendix, we report both abundance and growth rate, but in presenting criteria tend to focus on the growth rate targets. Expressing the target as a growth rate emphasizes the key parameter driving the extinction risk evaluation, which is productivity.

A.



B.

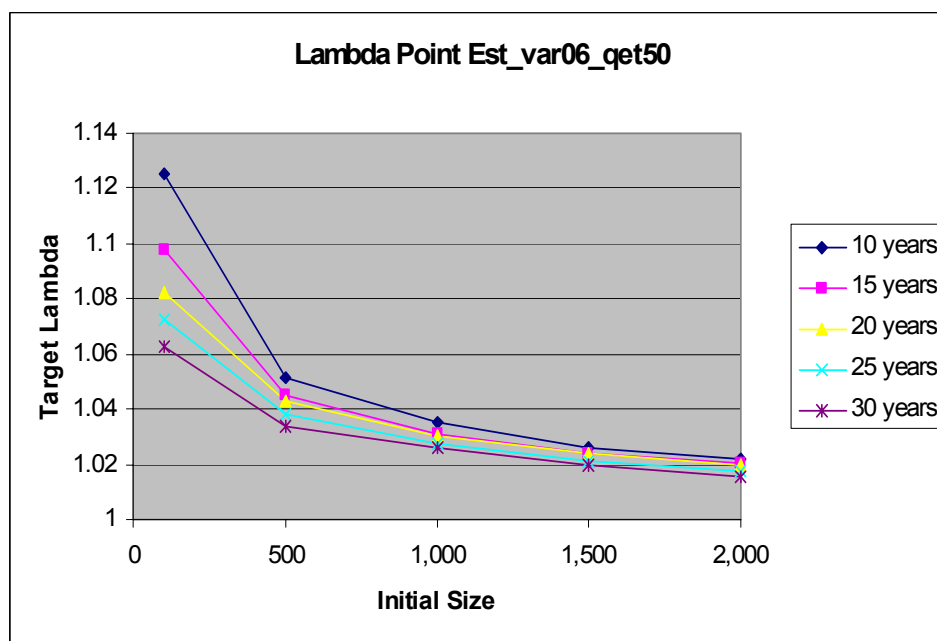


Figure D.7 Growth rates associated with the population change criteria in Figure D.6.

A computer program for calculating PCC based on user input is available at http://research.nwfsc.noaa.gov/cbd/trt/trt_wlc/viability_report.htm.

Parameter Uncertainty in Setting Criteria

There are a number of important assumptions and uncertainties associated with this approach to setting viability criteria. One major source of uncertainty is model uncertainty. Any model is a simplification of reality that attempts to capture the key elements of the problem in order to address specific questions. The appropriateness of the model construct we have used is discussed in the section “Model Uncertainty.” In this section, we discuss incorporating uncertainty surrounding parameter estimation in the criteria. In applying the criteria, three parameters are estimated from time series of abundance: σ^2 , $\gamma (= \lambda)$, and population abundance. The other biologically informed model parameter, QET, is not estimated from the salmon time series.

Because there is natural variability in the system and only relatively short time series are available, there is some probability that the point estimates generated for σ^2 and λ will not reflect the true parameter values. This uncertainty is captured in the parameters’ sampling distributions. The sampling distributions of σ^2 and λ can be estimated based on the model assumption that $\ln(N_{t+1}/N_t)$ is normally distributed. The sampling distribution of σ^2 is given in Equation 4 and is a function of the point estimate of the variance, $\hat{\sigma}^2$, and the degrees of freedom for the estimate, which is a direct function of the number of years of data used to calculate the variance estimate. The sampling distribution of λ is:

$$\lambda \approx e^{\mu},$$

$$\mu \approx \hat{\mu} - \text{tin}v(df) \sqrt{\frac{\hat{\sigma}^2}{b}}, \quad \text{Eq. 6}$$

$$\hat{\mu} = \text{mean} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right), \quad \text{Eq. 7}$$

$$\hat{\sigma}^2 = \text{var} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right), \quad \text{Eq. 8}$$

where $\text{tin}v(df)$ is the inverse t-distribution with df degrees of freedom, df is the degrees of freedom associated with the variance estimate, and b is the number of N_{t+1}/N_t ratios used to calculate $\hat{\mu}$. If the four-year running sum approach is used, b = number of years of spawner counts minus 4. Note that the time series used to estimate $\hat{\sigma}^2$, does not need to be identical to the time series used to estimate $\hat{\mu}$, and the df associated with the sampling distribution is functionally independent of the b parameter. This allows the use of the variance estimate and degrees of freedom associated with the pooled variance estimate in determining the sampling distribution of λ (see Appendix E). The b parameter will be a function of the number of years needed to achieve the target.

Because there is uncertainty in the parameter estimates, the true probability of extinction is not simply the fraction of time the population with point estimate σ^2 and γ values is expected to go extinct. There is some probability that the true σ^2 value is higher than $\hat{\sigma}^2$ and/or that the true μ is lower than $\hat{\mu}$, in which case the probability of extinction would be higher than that

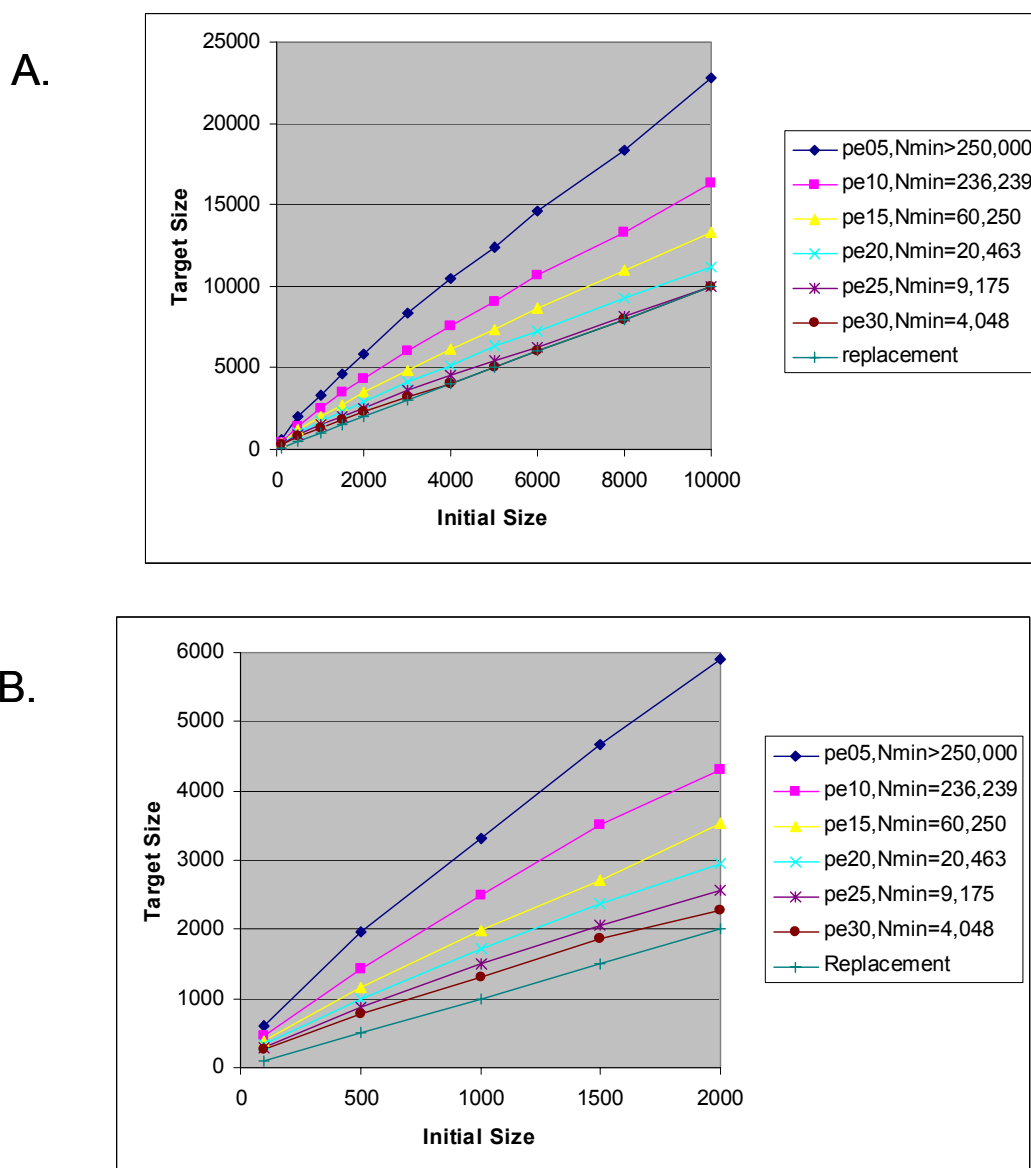
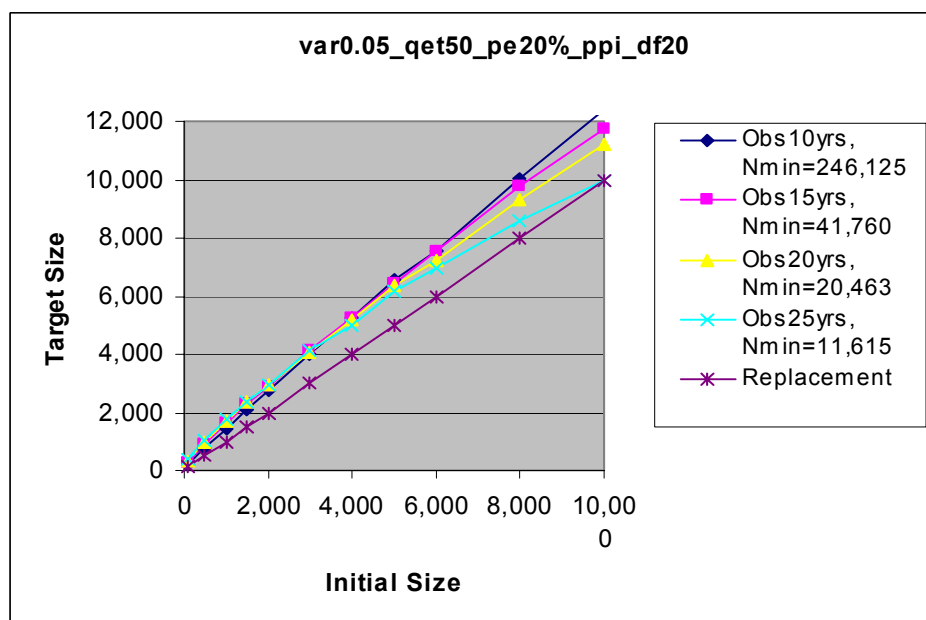


Figure D.8 Population growth criteria based on population prediction intervals. The point estimate of σ^2 is 0.05. The degrees of freedom for the variance estimate was given as 20. The different curves represent different probabilities of declining to a four-year average of 50 spawners in 100 years. The time to reach the target size is fixed at 20 years. The Nmin values in the figure key show the abundance at which the target size is equivalent to the initial size. For any abundance above this Nmin value, the population simply needs to show the same four-year average abundance after 20 years as the initial size. Panel B shows an expansion of the lower portion of the x axis of panel A. The “replacement” curve is for reference purposes; it indicates where the target size equals the initial size.

estimated by the parameter point estimates. Likewise, there is some probability that the true σ^2 value is lower than $\hat{\sigma}^2$ and/or that the true μ is higher than $\hat{\mu}$, in which case the probability of extinction would be lower than that estimated by the parameter point estimates. To account for this uncertainty, we calculated the population prediction intervals to establish the PCC targets (Figures D.8 through D.10).

A.



B.

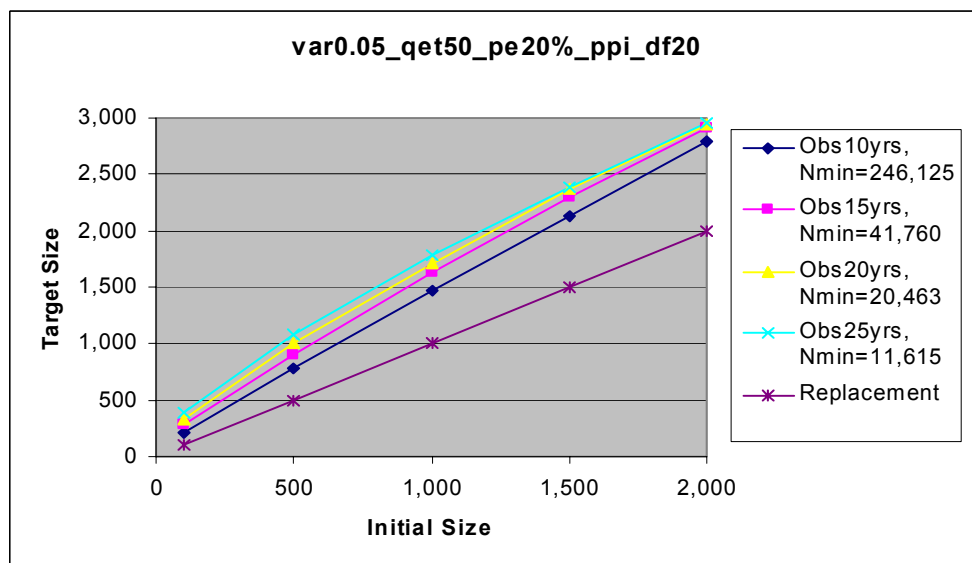


Figure D.9 Population change criteria showing the effect of different values of the time to reach the target. The criteria are based on population prediction intervals. The variance is 0.05 with 20 degrees of freedom, and the acceptable risk is a 20% probability of declining to a four-year average of 50 spawners in 100 years. Panel B shows an expansion of the lower portion of the x axis of panel A. The “replacement” curve is for reference purposes; it indicates where the target size equals the initial size.

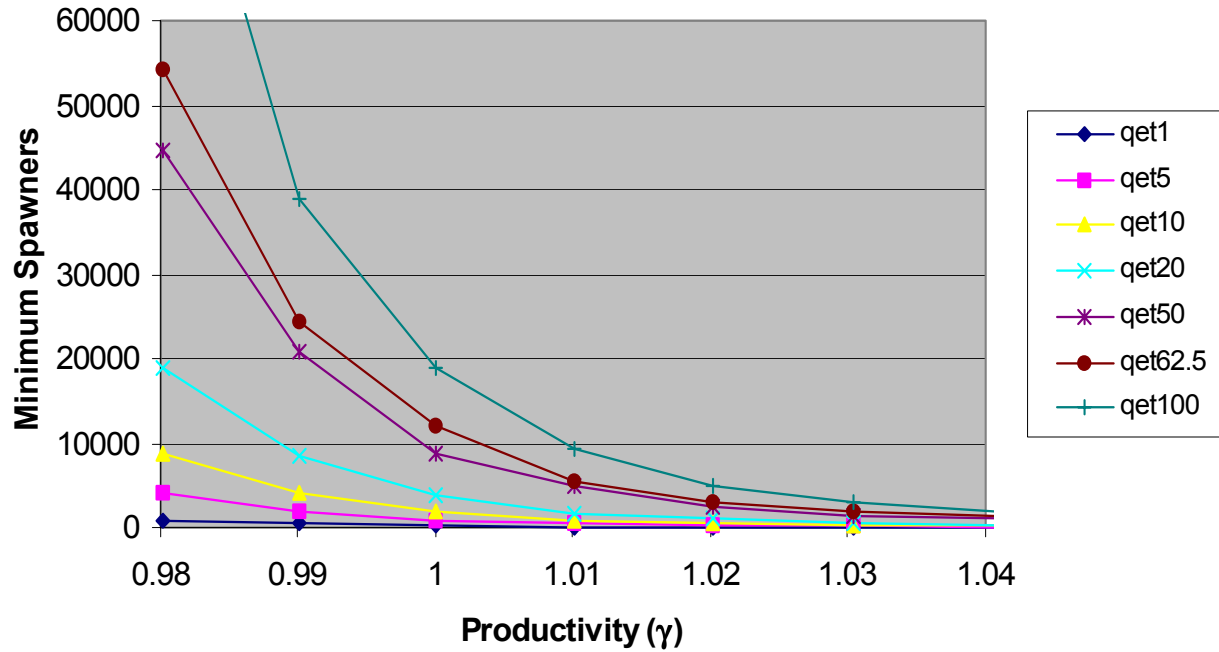


Figure D.10 Point estimates of λ associate with reaching the PPC in Figure D.9.

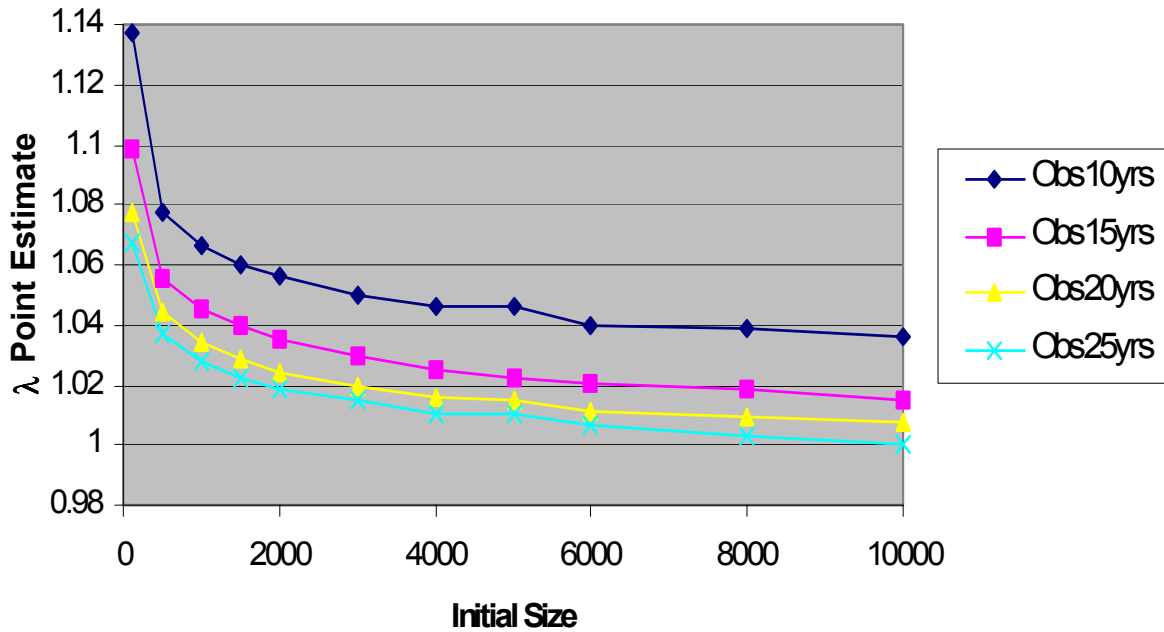
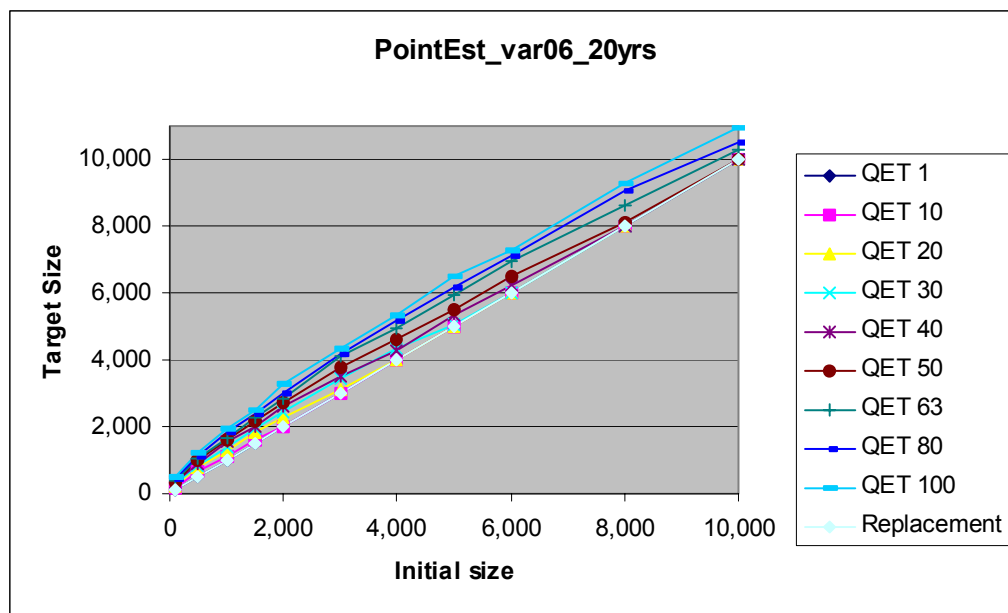


Figure D.11 Viability curves for different values of QET. The variance is 0.06, and the acceptable risk is a 5% probability of declining to a four-year average of QET spawners in 100 years. Note that as the productivity increases, the difference in minimum size associated with different QET values decreases.

A.



B.

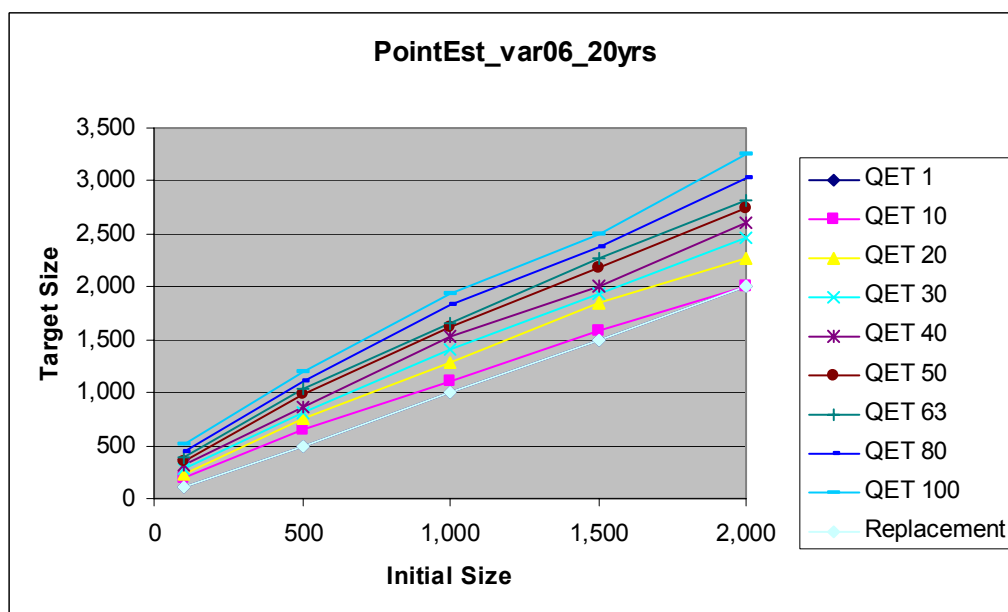


Figure D.12 Population change criteria showing for different values of QET. The criteria are based on point estimates. The variance is 0.06 and the acceptable risk is a 5% probability of declining to a four-year average of QET spawners in 100 years. The time to reach the target is fixed at 20 years. Panel B shows an expansion of the lower portion of the x axis of panel A. The “replacement” curve is for reference purposes; it indicates where the target size equals the initial size.

trajectories that drop below QET within the specified period of time (e.g., 100 years). Rather than parameterize the model simply using the point estimates, the γ and σ^2 parameters are drawn independently and randomly from the appropriate sampling distributions. This approach has been referred to in the literature as population prediction intervals, parametric bootstrapping, or simply

a type of Monte Carlo simulation. Figures D.6 and D.8 compare extinction risks calculated with point estimates and risks calculated using population prediction intervals. When we incorporate the uncertainty associated with parameter estimation into our assessment of extinction risk, we generally require larger target population size for a given acceptable level of risk. Original guidance from NMFS identified an acceptable population extinction risk of a 5% probability of extinction in 100 years for a VSP.

In order to evaluate the status of a population relative to the criteria, it is also necessary to estimate its abundance at the initial and target time periods. The time series of abundance is not informative regarding the accuracy of the abundance estimates. To assess uncertainty about the abundance estimates, it is necessary to know something about the measurement and sampling error associated with the count method. The WLC-TRT has not yet evaluated the errors associated with different abundance estimates; we assume that the initial and target abundances are measured precisely and without bias. As future studies evaluate the accuracy of abundance counts, the target sizes may need to be adjusted to achieve the same level of certainty about the population extinction risk.

The QET is a biological parameter that is not estimated from salmon data. The only way we can incorporate uncertainty about QET into our criteria assessment is through sensitivity analysis (Figures D.11 and D.12). In sensitivity analysis, we explore the effect of changing the assumption about QET on the proposed criteria. As the γ value increases, the effect of QET declines.

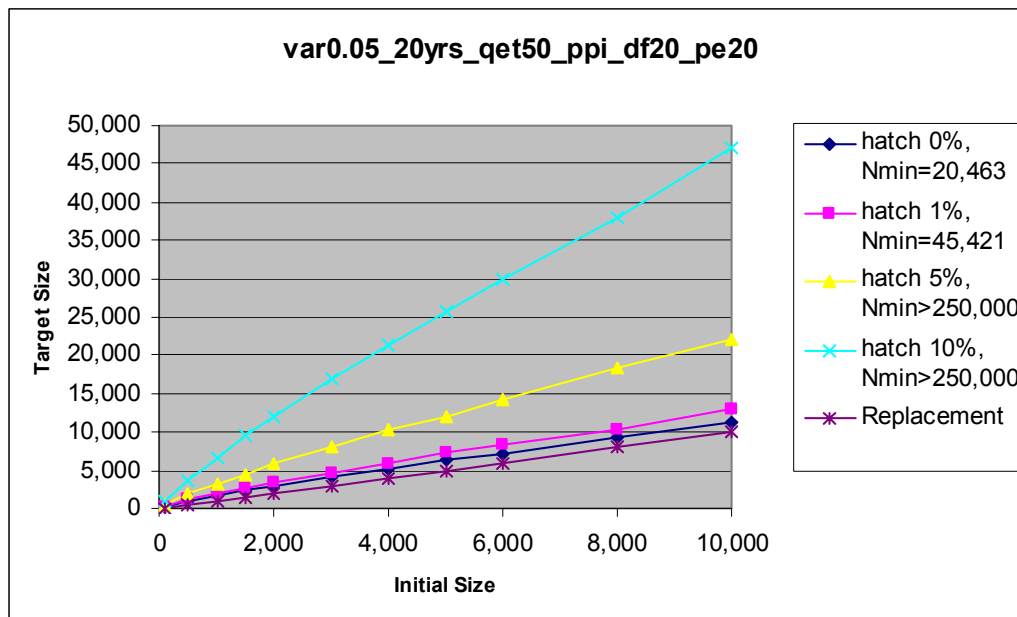
Hatcheries and PCC Targets

In assessing viability, we are concerned with the question of whether a population would be naturally self-sustaining. Hatchery-origin fish that spawn with natural-origin fish have the potential to “mask” the productivity of the wild population (McClure et al. in review). The equation for estimating the growth rate used to calculate the PCC target of a population with hatchery-origin fish is:

$$\hat{\lambda} = \exp \left(\text{mean} \left(\ln \left(\frac{N_{t+1}}{N_t + hN_t} \right) \right) \right) = \exp \left(\frac{\ln \left(\frac{\phi}{t} * (1 - \eta)^b \right)}{y} \right), \quad \text{Eq. 9}$$

where N_t is the number of natural-origin spawners in year t , hN_t describes the effective number of hatchery-origin fish spawning in year t as a function of N_t , ϕ is the target number of natural-origin spawners, t is the current number of natural-origin spawners, η is the effective proportion of the spawning population of hatchery origin, and y is the number of years between observations. The effective proportion of hatchery-origin spawners may be different from the census count proportion of hatchery-origin spawners if hatchery-origin fish have a different reproductive success than natural-origin spawners. The fraction of hatchery-origin spawners is the fraction anticipated over the target period. Figure D.13 shows the effect of changing the fraction of hatchery-origin spawners. A relatively small fraction of hatchery-origin spawners can have a big impact on the target size needed to demonstrate a given level of productivity. To

A.



B.

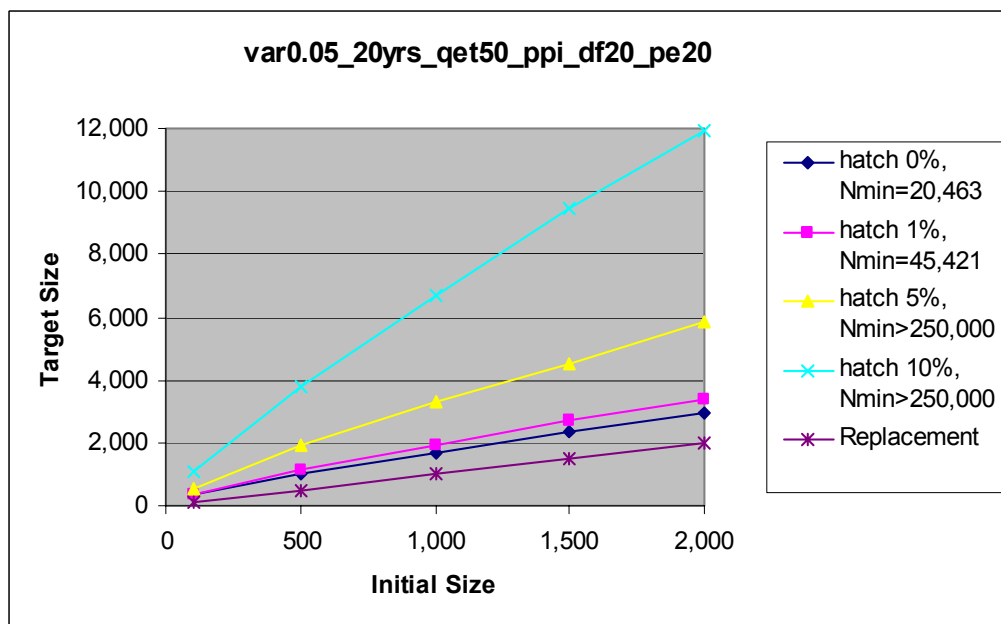


Figure D.13 Population change criteria showing the effect of different fractions of hatchery-origin spawners. The criteria are based on population prediction intervals. The variance is 0.05 with 20 degrees of freedom and the acceptable risk is a 20% probability of declining to a four-year average of 50 spawners in 100 years. Panel B shows an expansion of the lower portion of the x-axis of panel A. The “replacement” curve is for reference purposes and indicates where the target size equals the initial size.

evaluate the productivity of a population with hatchery-origin spawners, it is necessary to have an accurate estimate of the effective fraction of hatchery-origin fish.

Ocean Cycles

The population dynamics model described in Equation 1 assumes no temporal autocorrelation in productivity. However, salmon are recognized as experiencing decade-scale periods of higher- or lower-than-average productivity as a result of long-term cycles in ocean conditions (Mantua et al. 1997, Anderson 1998, Beamish et al. 1999, Hare et al. 1999). These long-period “regime shifts” are difficult to model because they are difficult to predict. However, they can have significant consequences for setting and evaluating performance of viability criteria. It is important to not conclude that population is viable during a period of high marine survival if it can be anticipated that the population is likely to go extinct during the next period of low marine survival. Likewise, we would not want to conclude that a population is not viable during a period of low ocean survival if it can be anticipated that the long-term prospects for the population are good, given that it is likely to soon enter a period of higher ocean survival. These issues are illustrated in Figure D.14. We partially address this concern about ocean cycles by including juvenile outmigrant (JOM) criteria, which attempt to separate out the freshwater and marine survivals. However, we also considered marine cycles in setting adult abundance viability criteria.

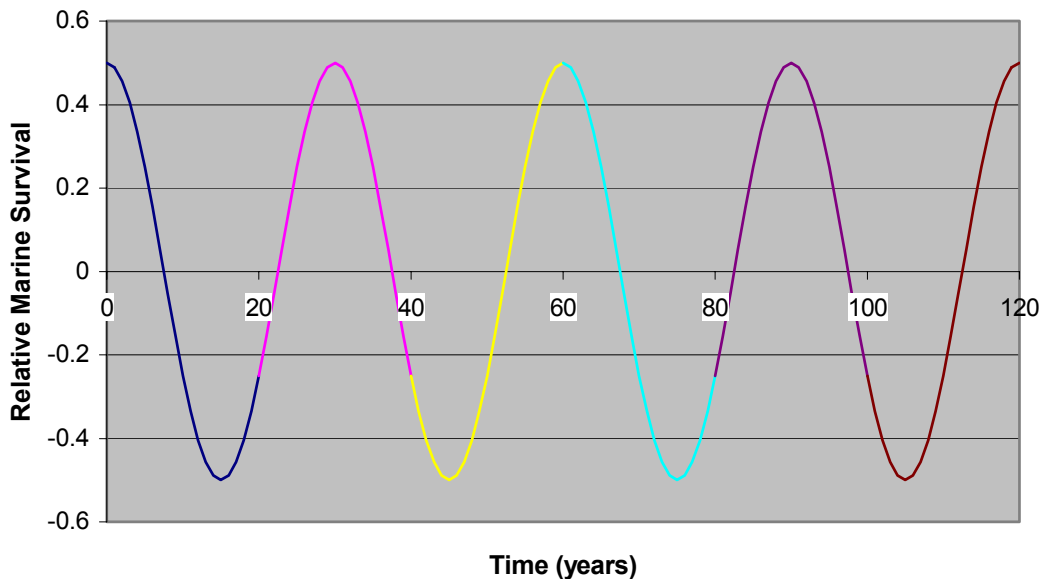


Figure D.14 Conceptual graph of 30-year marine survival cycles. Different colors in the curve represent different potential periods over which the target is achieved. Each potential observation period would have different marine index ratios. Real marine survival patterns are not nearly as predictable as this sine wave.

Given that it is difficult to predict patterns of marine survival, we took the approach of modifying the target criteria as a function of how the marine survival over the target period compared to the long-term average marine survival (Figure D.15). The modification, applied to the calculation of λ over the target period, is as follows:

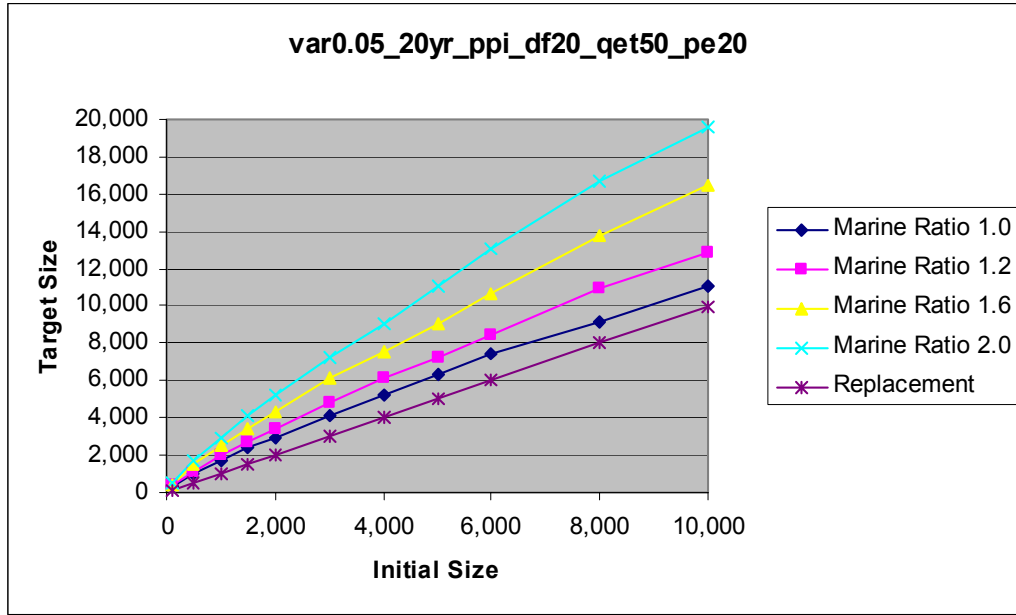
$$\hat{\lambda} = \exp \left(\frac{\ln \left(\frac{\phi}{t} \right)}{y} - \frac{\ln \left(\frac{\nu}{\theta} \right)}{y} \right), \quad \text{Eq. 10}$$

where ν is the marine survival index observed over the target period, θ is the long-term average value of the same marine survival index and all other symbols are as in Equation 5. A basic assumption of this approach is that the target values calculated without the correction represent the minimum sizes based on some long-term average growth rate. When we apply the correction, we assume that the observed growth rate differs from the long-term average growth rate in an amount that is proportional to the difference between the observed marine index and the long-term average marine index. Since there is logically a direct relationship between ocean survival and productivity throughout the life cycle, this a reasonable assumption.

In developing the viability criteria, we applied this correction asymmetrically; that is, the modification is only used to increase the target during periods of high ocean survival, not to reduce the target during periods of low ocean survival. This is a precautionary application. If we observe a population with a marine survival over the target period that is higher than long-term average, we are relatively certain that at some future time the marine survival will decrease; thus we should stipulate a higher target during the “good” ocean years. The converse is not necessarily true. If we observe a lower than long-term average marine survival over the target period, it is not clear that marine survival will improve. This is because human activities—such as those that affect global warming—may have permanently reduced ocean productivity for salmon, or the condition of fish as they leave freshwater may be the cause of the low marine survivals. For these reasons, we did not lower the abundance target during periods of low ocean survival.

We have not yet identified the appropriate index (assuming one exists) to use for this marine survival modification to the target criteria. Several candidates exist, including measures of marine survival estimates from hatchery-marked fish or physical indexes such as the Pacific Decadal Oscillation (PDO) or El Niño-Southern Oscillation (ENSO), which are correlated with salmon marine survival. Although many features of this marine index approach are conceptually attractive, whether it can be satisfactorily implemented remains to be seen.

A.



B.

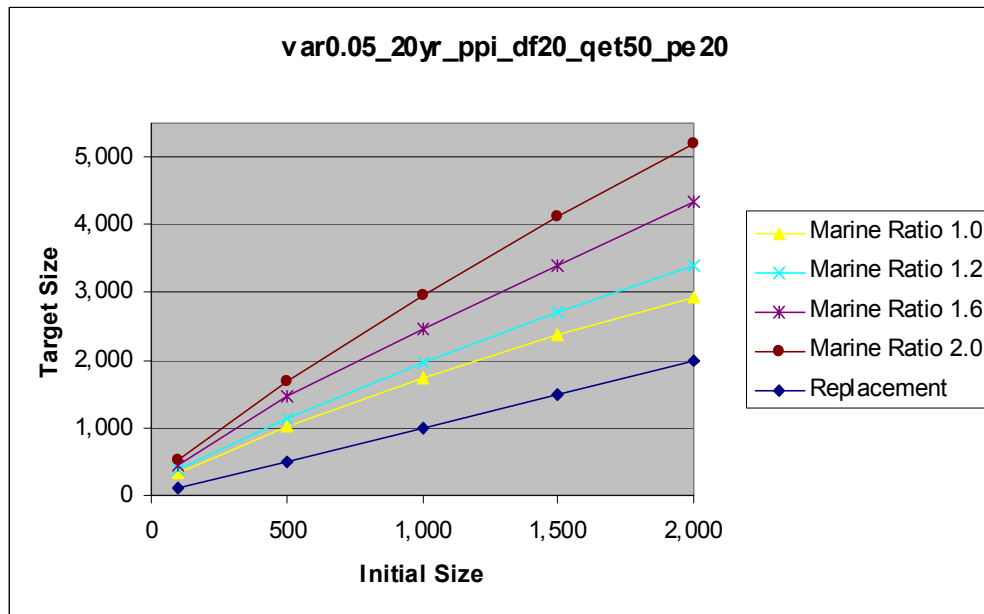


Figure D.15 Population change criteria showing the effect of marine survival modification. The marine ratio is the marine survival index observed over the target period divided by the long-term average marine index. The criteria are based on population prediction intervals. The variance is 0.05, with 20 degrees of freedom, and the acceptable risk is a 20% probability of declining to a four-year average of 50 spawners in 100 years. Panel B shows an expansion of the lower portion of the x axis of panel A. The “replacement” curve is for reference purposes; it indicates where the target size equals the initial size.

Model Uncertainty

We address model uncertainty by evaluating how well the criteria performed when confronted with simulated time series abundance data that was generated using processes other than those used to set the criteria (McElhany and Payne in prep.) (Figure D.16). For example, we generated a large number of trajectories with different recruitment functions (e.g., Ricker, Beverton-Holt), short-lag autocorrelations, decadal-scale regime shifts, and changes in population carrying capacity. We then calculated viability criteria using the early part of the simulated time series, determined the conclusion we would reach about the population after applying the criteria to the next segment of the time series, and finally looked at the long-term fate of the simulated population to determine whether our conclusions were correct. For every scenario tested we generated a table like Table D.2 to examine the rate at which the criteria lead to certain types of errors. The criteria tested by McElhany and Payne are not identical to the

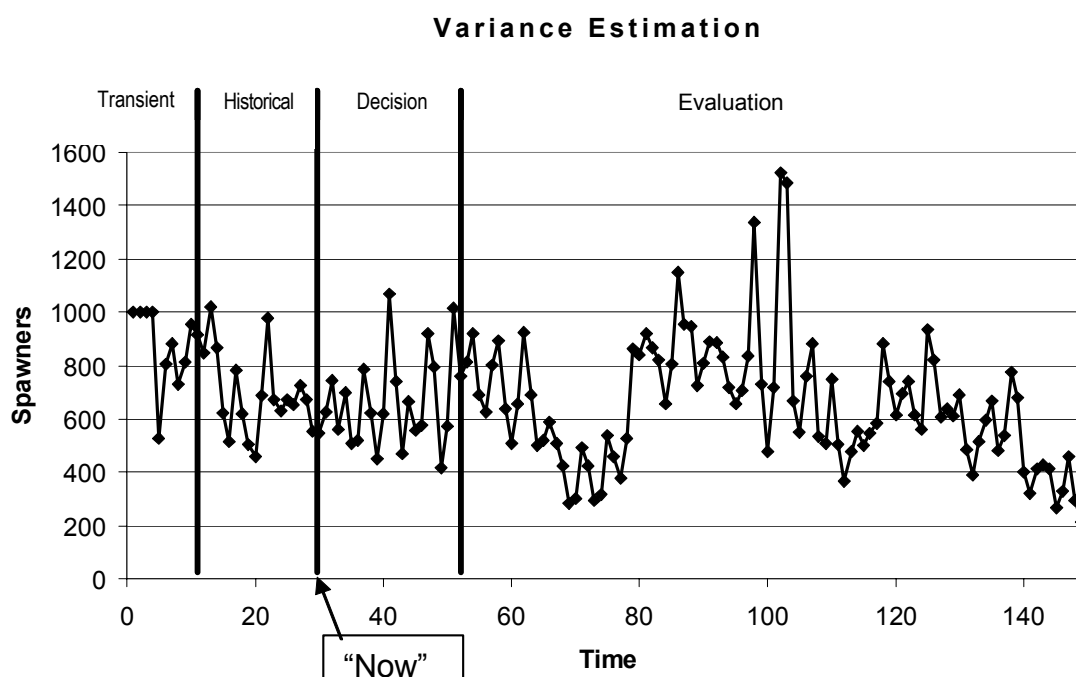


Figure D.16 Sample trajectory illustrating the approach used to evaluate the viability criteria showing variance estimation, decision, and evaluation period. The first 10 years, during which initial transients in the age structure were allowed to stabilize, was not used for estimation or evaluation. The variance estimation period was used to estimate process variance and set the viability curve. The variance estimation period overlapped with the decision period. In most of our simulations, we assumed that it included a period of historical data and was updated to include data from the decision period. The decision period was used to estimate the growth rate and reach a decision about whether or not to delist the population. The evaluation period was used to explore the fate of the simulated population after the delisting decision was made.

Table D.2 Possible outcomes of criteria applied to simulated trajectories.

Population fate	Delisting Decision	
	Delist	Do Not Delist
	Extinct	Correct
	Not extinct	Type II error

criteria presented in this report (for example, the marine index modification is a recent addition to the criteria), but the criteria in the drafts are very similar, and the general conclusions are appropriate to both. In general, the criteria were robust to the exact function of the population dynamics model (e.g., Ricker versus hockey-stick recruitment function, presence of short-lag autocorrelation, etc.). As expected, the criteria lead to the wrong conclusion most often when the population is starting at carrying capacity and has a high intrinsic productivity. Under these conditions, a population has a relatively low risk of extinction, and the criteria tended to be overly precautionary by not recognizing the populations as viable. Given the low current abundance of most populations, it is anticipated that most populations will need to grow to be considered viable, and this overly precautionary scenario will be the exception rather than the rule.

Minimum Targets

The PPC approach is appropriate once the initial population size is above a certain level, but it does not work well at extremely small initial sizes. For example, we cannot use the approach to set a target for a currently extirpated population. The analysis requires evaluating the term $targetSize/initialSize$. Since $initialSize$ for an extirpated population is 0, the term is undefined, and no target size can be identified. Even if we have a non-zero initial size, so that the equations are solvable, there is still a difficulty at small population size. If the initial size is one fish and the population increases to 50 fish over 20 years, the growth rate for the population is large ($\lambda = 1.28$, or a 28% increase per year), and because of the large growth rate, a population size of 50 may exceed the minimum size requirement for an acceptable risk (this is a function of the variance and QET). However, 50 fish may not be considered adequate target abundance for a number of reasons. One primary reason is because the proportional error rates in abundance estimates tend to be higher at small abundance (Holmes and Fagan 2002). Therefore, an estimate of productivity made at small population size is more likely to be wrong than an estimate made at higher population size. Consequently, we developed a set of minimum targets that should be met no matter how low the initial estimate of abundance. These minimum targets are based on setting a minimum initial population size that will serve as the basis for target criteria for all populations starting below the minimum initial size. Because of the uncertainty concerns, we have explored a number of values as the minimum initial size. If a population is below the minimum default value and achieves the targets for a population with an initial size of the minimum default value, the population will actually have a higher point estimate productivity than would be required if the criteria algorithm were simply applied at the low abundance.

Alternative Methods of Estimating Productivity

The population change criteria provide a precautionary and statistically defensible approach to estimating the intrinsic productivity of a population. However, in some cases it may not be necessary to directly observe population growth in order to conclude that a population has a productivity-size combination with an acceptably low risk level. If a population demonstrates a productivity-size combination above the appropriate viability curve, the population would be considered viable.

As discussed in Appendix G, fitting recruitment models to abundance data generally provides poor estimates of intrinsic productivity, but in particular cases data may support the use of this method. Appendix H describes a particular two life-stage recruits-per-spawner model. Information available for harvested populations may provide additional data to evaluate the productivity of a population. Given certain assumptions about natural levels of post-harvest mortality, it may be possible to estimate something about the “resilience” of a population (though not necessarily its intrinsic productivity). Calculations involving harvest would need to have an accurate method of assessing the harvest rate actually experienced by a particular population. In addition, an accurate accounting of hatchery fish in the system would be required to estimate natural productivity.

To be used to evaluate the viability of a population, any alternative method of estimating population productivity would need to meet reasonable standards of statistical rigor. The potential use of alternative methods to estimate productivity does not really aid in specifying, *a priori*, a particular point on the viability curve to use as a target. Rather, the alternative methods may be used to retrospectively evaluate whether or not a population should be considered viable.

Application of Population Change Criteria to Healthy Populations

The PCC approach is only applicable for evaluating whether or not a population that has been depressed below its historical abundance has improved in status and should be considered viable. If a population has not been depressed below its historical abundance, it would not be expected to grow in the future. If a population is not growing, the PCC approach assumes that the population productivity is 1. Abundance targets associated with a productivity of 1 are often larger than estimates of historical abundance. We would intuitively categorize a population that is stable at about its historical abundance as “healthy” because we are assuming, perhaps unconsciously, that the population productivity is actually greater than 1, and that the population is not growing because it is constrained by carrying capacity. If a population is stable at about historical abundance, we may not require further evidence about its productivity to conclude that it is viable. Alternatively, we may be able to apply one of the alternative methods for estimating productivity described in the previous section.

Most Willamette/Lower Columbia (WLC) populations are substantially below historical abundance and are not considered currently healthy, hence the Endangered Species Act (ESA) listing. Even the most abundant population, the Lewis River bright chinook salmon population, at its most recent four-year annual average of 8,900 spawners, is well below the historical estimate of equilibrium abundance based on habitat productivity viability analysis (HPVA) of 43,000 spawners. Even given the uncertainties associated with the ecosystem diagnosis and

treatment (EDT) estimates, it seems likely that there is, at least theoretically, potential for the population to grow.

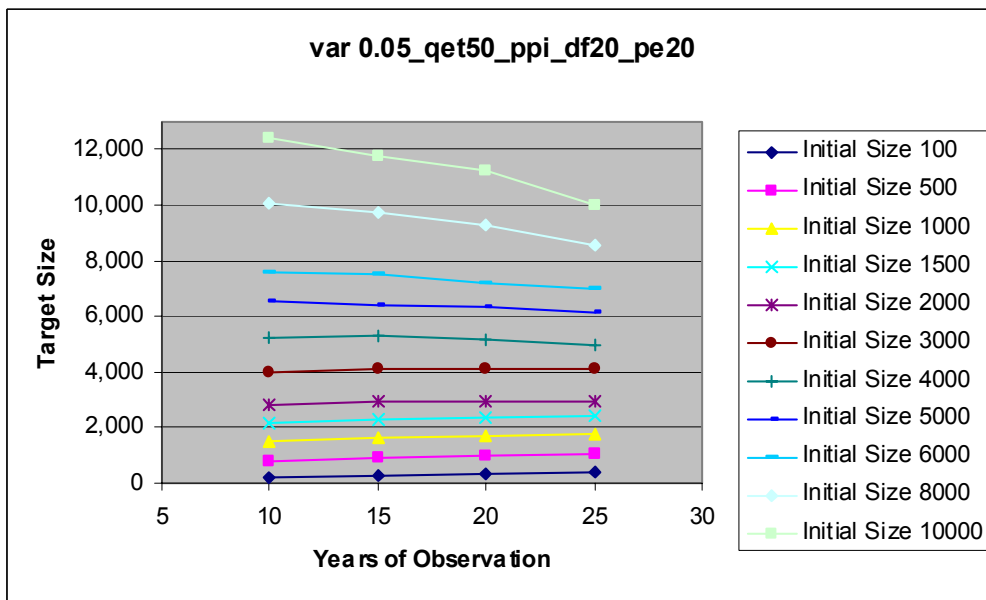
Evaluation Time Period

Power analyses indicate that at least 12 years of data are required before λ estimates have any meaning (Holmes 2001, Holmes and Fagan 2002, McElhany and Payne in prep, McClure et al. 2003). While we have shown 10-year observation periods for illustration purposes, 10 years is really too short; 15 to 20 years is more appropriate, both in terms of estimating growth rate and averaging over a longer portion of any marine survival cycles (Figures D.6, D.7, and D.17). However tempting it may be to conclude that a population is okay if it achieves the target abundance before 15 to 20 years, it is crucial to recognize that such a conclusion would be statistically unsound. The criteria are based on variability patterns, and it is necessary to wait and see if the population is still above the target size after the target time. Even a declining population may momentarily exceed the target size, and it is the long-term behavior of the population that is relevant.

An important question in applying these criteria is when to start evaluating population status. One strategy is to simply start with the current population size and look forward. Alternatively, we can stipulate that any time series of acceptable length that meets the criteria and includes the most recent year's data would qualify as viable. While the later option may be possible in some populations, for many of them there is simply no credible historical time series available: starting from the present and looking forward is the only option. Given the sensitivity of the criteria to small changes in the fraction of hatchery-origin spawners, it becomes even more unlikely that historical data are adequate. However, it is possible to include data before 2002 in assessing the status of populations if the data are of sufficient quality.

It is not possible to entirely stipulate the criteria in advance because they depend on evaluating marine survivals over some future period. Although the projected fraction of hatchery-origin spawners can be estimated, it too will need to be actually evaluated to determine if the abundance target is adequate. As part of an adaptive management protocol, the variance estimates should also be updated as more data become available.

A.



B.

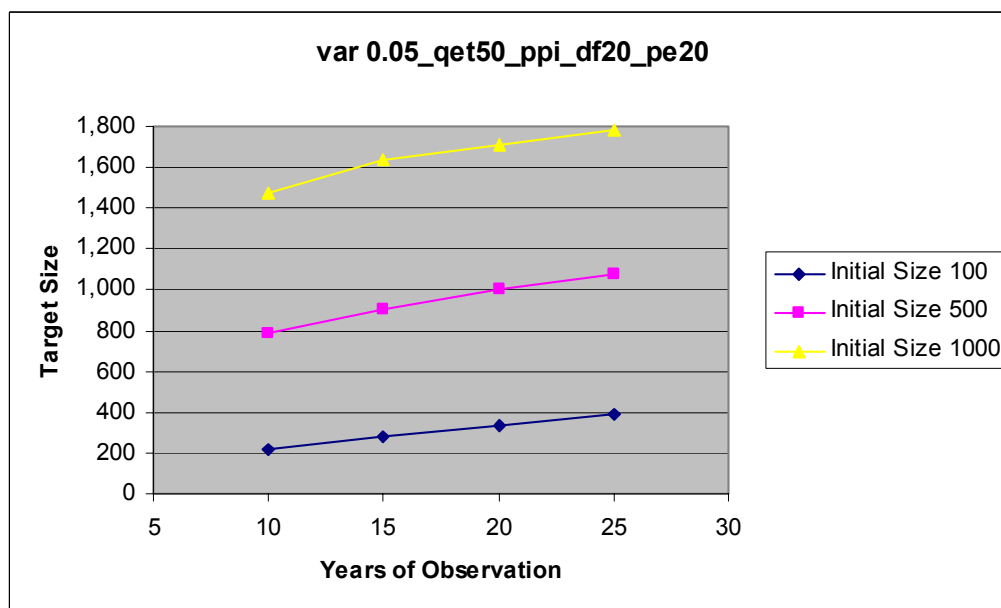


Figure D.17 Target size as a function of the number of years to reach the target for a number of different initial population sizes. The criteria are based on population prediction intervals. The variance is 0.05 with 20 degrees of freedom, and the acceptable risk is a 20% probability of declining to a four-year average of 50 spawners in 100 years. Panel B shows an expansion of the lower portion of the y axis of panel A.

PCC Criteria in the WLC

Current Abundance and Hatchery Fraction

PCC targets (either growth rate or abundance) assume a variety of conditions, which can be found in Tables D.3 and D.4. An appropriate target could be determined from Table D.3 or D.4 if the current population size (Table D.5) and the other model parameters are known. The current population sizes for many WLC populations are found in Table D.5. The table also contains the recent fraction of hatchery-origin spawners for some populations, which could be used in conjunction with Table D.3, assuming that the current fraction of hatchery-origin spawners will continue into the future. However, hatchery production is under human control, and the future fraction of hatchery-origin spawners will reflect future policy decisions.

Variance Estimates

The key empirical parameter for setting the criteria is the estimate of environmental variance. Variance estimates for populations in the WLC domain are summarized in Appendix E. The Lower Columbia ESUs have average variance point estimates of about 0.05; a value of 0.05 was used to generate criteria for these populations. In general, the variance estimates (and targets) will need to be evaluated as more data become available.

Final PCC Recommendations

This appendix is intended to describe and illustrate the PCC approach by example. The final WLC-TRT recommendations regarding the PCC criteria are located in the main text of this document. The final recommendations include a discussion of when it would be appropriate to use the PCC approach as viability criteria and when other methods should be used.

Table D.3 Sensitivity analysis of PCC targets. Targets are expressed as observed, median, annual population growth rates, assessed on a four-year running sum.

Current Size		Standard ^a	Time Period 40 ^b	Variance Degrees of Freedom ^d										QET ^f			Extinction Risk ^g			Acceptable Time Horizon	Marine Index Long- Term ⁱ
				Variance ^c		Freedom ^d		Hatchery Fraction ^c		100	1	25	60	200 ^h							
				.01	.1	5	10	40	5%							10%	30%				
100	12%	7%	4%	18%	16%	13%	11%	16%	>21%	>21%	14%	16%	7%	4%	13%	13%					
150	11%	6%	4%	17%	15%	12%	10%	15%	20%	>21%	13%	15%	6%	2%	12%	12%					
200	11%	6%	3%	16%	15%	12%	10%	15%	20%	>21%	12%	14%	5%	1%	12%	12%					
500	9%	5%	3%	14%	12%	10%	9%	13%	19%	>21%	10%	13%	4%	-1%	11%	10%					
1,000	8%	4%	2%	13%	11%	9%	8%	12%	18%	>21%	9%	12%	2%	-2%	10%	10%					
1,500	7%	4%	1%	12%	10%	8%	7%	12%	17%	>21%	8%	12%	2%	-3%	9%	9%					
2,000	7%	4%	1%	12%	10%	7%	7%	12%	17%	>21%	8%	12%	2%	-3%	9%	8%					
2,500	7%	3%	1%	12%	10%	8%	6%	11%	16%	>21%	8%	11%	2%	-3%	9%	9%					
3,000	6%	3%	1%	12%	9%	7%	6%	11%	16%	>21%	7%	11%	1%	-3%	9%	8%					
3,500	7%	3%	0%	11%	9%	7%	6%	11%	16%	>21%	7%	10%	1%	-4%	9%	8%					
4,000	6%	3%	0%	11%	9%	7%	6%	11%	16%	>21%	7%	11%	1%	-4%	9%	8%					
4,500	6%	3%	0%	11%	9%	7%	6%	11%	16%	>21%	7%	10%	1%	-4%	9%	8%					
5,000	6%	3%	0%	11%	8%	7%	6%	11%	16%	>21%	7%	10%	1%	-4%	9%	8%					
6,000	6%	3%	0%	11%	9%	6%	5%	10%	16%	>21%	7%	10%	0%	-4%	9%	7%					
7,000	6%	3%	0%	11%	8%	6%	5%	10%	15%	>21%	6%	10%	0%	-5%	8%	7%					
8,000	5%	2%	0%	10%	8%	6%	5%	10%	15%	>21%	6%	11%	0%	-5%	8%	7%					
9,000	5%	2%	0%	10%	8%	6%	5%	10%	15%	>21%	6%	10%	0%	-5%	8%	7%					
10,000	5%	2%	-1%	10%	8%	6%	5%	10%	15%	>21%	6%	10%	0%	-5%	8%	7%					

^a This column describes the targets assuming standard conditions: for these analyses, they were a 20-year observation period, process variance of 0.05, 20 degrees of freedom for the variance estimate, 0 hatchery-origin spawners, a QET four-year average of 50 spawners per year, and an acceptable extinction risk of 5% in 100 years. The other target columns show target calculated by varying one of the standard assumptions and keeping all others the same.

^b Time Period 40 assumes the observation period is 40 years.

^c Variance 0.01 and 0.1 assume difference process variance values.

^d Variance Degrees of Freedom columns assume different variance degrees of freedom values.

- ^e Hatchery Fraction columns assume different fractions of hatchery-origin spawners in the population.
- ^f QET 100 shows targets assuming a QET of a four-year average of 100 spawners per year.
- ^g Extinction Risk columns assume an acceptable extinction risk of # percent in 100 years.
- ^h Acceptable Time Horizon 200 assumes an acceptable extinction risk of 5% in 200 years.
- ⁱ Marine Index Long-Term assumes the marine survival over the observation period was twice the long-term average.

Table D.4 Identical to Table D.3, except the targets are expressed as observed four-year average spawner abundances.

															Acceptable	Marine
Current	Standard ^a	Time Period 40 ^b	Variance Degrees of								QET ^f	Extinction Risk ^g			Time Horizon 200 ^h	Index Long-Term ⁱ
			Variance ^c		Freedom ^d			Hatchery Fraction ^e								
Size			.01	.1	5	10	40	5%	10%	30%	100	1	25	60		
100	600	1,200	200	1,400	1,100	700	500	1,060	>2,000	>2,000	800	1,000	300	200	700	700
150	800	1,400	300	1,800	1,500	900	700	1,459	2,797	>3,000	1,000	1,400	400	200	1,000	1,000
200	1,000	1,700	300	2,100	1,800	1,200	900	1,835	3,754	>4,000	1,200	1,700	500	200	1,200	1,200
500	1,900	3,000	700	4,300	3,200	2,200	1,900	3,613	7,618	>10,000	2,300	3,600	900	400	2,500	2,500
1,000	3,400	4,600	1,300	7,400	5,400	3,800	3,200	6,283	13,768	>20,000	3,900	6,500	1,500	700	4,600	4,400
1,500	4,700	6,000	1,900	9,600	7,400	5,400	4,500	8,938	19,358	>30,000	5,400	9,000	2,100	1,000	6,200	6,000
2,000	6,000	7,200	2,300	12,200	9,000	6,300	5,500	11,721	23,737	>40,000	6,800	12,000	2,600	1,200	8,000	7,100
2,500	7,100	8,500	2,800	14,700	11,100	8,200	6,800	14,191	28,397	>50,000	8,100	13,600	3,200	1,400	10,100	9,400
3,000	8,200	9,900	3,300	17,100	12,700	9,000	7,800	16,699	33,955	>60,000	9,400	15,600	3,600	1,700	11,700	10,300
3,500	9,700	10,900	3,800	19,200	14,800	11,000	9,000	18,349	39,406	>70,000	10,500	17,000	4,100	1,900	13,500	11,700
4,000	10,600	11,300	4,300	21,700	16,000	11,300	10,100	21,297	42,670	>80,000	11,300	20,200	4,600	2,100	14,900	13,800
4,500	11,400	12,500	4,600	24,000	17,500	12,800	10,800	23,032	47,254	>90,000	12,800	21,800	5,100	2,300	16,800	14,600
5,000	12,800	14,500	5,100	25,000	18,400	14,500	12,100	24,806	51,380	>100,000	14,600	23,400	5,500	2,500	18,800	15,900
6,000	14,800	15,900	6,000	30,300	22,800	16,100	14,100	29,057	61,153	>120,000	17,000	28,800	6,100	2,900	22,300	18,300
7,000	17,200	17,400	6,800	35,100	24,700	18,100	15,900	32,254	69,359	>140,000	18,800	32,100	7,300	3,300	25,500	21,900
8,000	17,500	18,700	7,700	36,900	27,900	20,600	18,300	37,051	80,045	>160,000	20,800	40,100	8,200	3,800	28,000	25,100
9,000	20,900	21,700	8,500	40,700	30,800	22,700	20,100	39,393	85,742	>180,000	23,400	39,600	8,900	4,100	32,100	25,900
10,000	21,700	23,600	9,200	45,100	34,400	24,700	21,900	45,669	93,802	>200,000	25,300	43,200	9,500	4,600	34,800	28,400

^a This column describes the targets assuming standard conditions: for these analyses, they were a 20-year observation period, process variance of 0.05, 20 degrees of freedom for the variance estimate, 0 hatchery-origin spawners, a QET four-year average of 50 spawners per year, and an acceptable extinction risk of 5% in 100 years. The other target columns show target calculated by varying one of the standard assumptions and keeping all others the same.

^b Time Period 40 assumes the observation period is 40 years.

^c Variance 0.01 and 0.1 assume difference process variance values.

^d Variance Degrees of Freedom columns assume different variance degrees of freedom values.

^e Hatchery Fraction columns assume different fractions of hatchery-origin spawners in the population.

^f QET 100 shows targets assuming a QET of a four-year average of 100 spawners per year.

^g Extinction Risk columns assume an acceptable extinction risk of # percent in 100 years.

^h Acceptable Time Horizon 200 assumes an acceptable extinction risk of 5% in 200 years.

ⁱ Marine Index Long-Term assumes the marine survival over the observation period was twice the long-term average.

Willamette/Lower Columbia Salmonid Viability Criteria

Table D.5 Recent average abundance and fraction of hatchery origin for WLC populations.^a

ESU	Population ^b	Year	Current Size	Hatchery Fraction
Columbia chum salmon	Grays River	1997–1998	874	0
	Lower gorge tributaries	1997–2000	542	0
	Upper gorge tributaries	1997–2000	100	
Upper Willamette steelhead	Mollala River	1997	574	24
	North Santiam River	1997	2,214	29
	South Santiam River	1997	900	0
	Calapooia River	1997	236	0
Upper Willamette chinook salmon	Clackamas River	1997–2000	1,453	
	McKenzie	1997–2000	1,904	24
Lower Columbia steelhead	North Fork Toutle River winter	1997–2000	176	0
	South Fork Toutle River winter	1997–2000	463	2
	Coweeman River winter	1998–2000	487	50
	Kalama River winter	1997–2000	554	0
	Clackamas River winter	1997–2000	465	39
	Sandy River winter	1997–2000	1,005	
	Hood River winter	1997–2000	850	52
	Kalama River summer	1997–2000	419	38
	East Fork Lewis summer	1997–2000	287	33
	Washougal River summer	1997–2000	158	8
	Wind River summer	1997–2000	368	10
	Hood River summer	1997–2000	866	82
Lower Columbia chinook salmon	Grays River fall	1997–2000	127	37
	Elochoman River fall	1997–2000	754	69
	Mill, etc. fall	1997–2000	491	47
	Lower Cowlitz fall	1997–2000	1,702	67
	Coweeman	1997–2000	425	0
	Kalama River fall	1997–2000	2,995	67
	Salmon Creek late fall	1997–2000	235	0
	Washougal River fall	1997–2000	3,231	57
	Sandy River fall	1997–2000	220	3
	Upper gorge tributaries fall	1997–2000	159	17
	Big White Salmon fall	1997–2000	234	21
	Sandy late	1997–2000	839	3
	North Fork Lewis bright	1997–2000	7,293	13
	Upper Cowlitz spring	1997–1999	365	
	Kalama River spring	1997–1999	105	0
	Lewis River spring	1997–1999	300	0

^a The averages are standardized for the years 1997–2020: if data were missing over these years, the average was based on the existing data.

^b This list does not include all WLC populations. Some populations are extirpated and have a current abundance of 0. For populations not in this table, there are no available abundance data.

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APPENDIX E

ESTIMATES OF ENVIRONMENTAL VARIANCE FOR PCC ANALYSIS

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Overview

Environmental variance is a key parameter in the extinction risk model used to set population change criteria (PCC). The basic approach we used to estimate environmental variance is the slope method, described in Holmes (2001). This method helps correct for the large upward bias in the variance estimate that is produced by measurement error. The basic equation of the slope method is:

$$\hat{\sigma}^2 = \text{slope of } \text{var} \left(\ln \left(\frac{N_{t+\tau}}{N_t} \right) \right) \text{ vs. } \tau ,$$

where N_t is a running sum of spawner abundance counts, and τ is the temporal lag between the values used for the variance estimate. For our variance estimations, we used a running sum of four years and estimated the slope based on maximum τ of 4, as did McClure et al. (2003).

In estimating extinction risk, we need to know the natural variance, because it affects the populations no matter what human actions are taken. The presence of hatchery-origin spawners can complicate the effort to determine natural variability because changes in hatchery output can uncouple observations of spawner abundance and natural population dynamics. To correct for this potential problem, we explored modifying the equation to estimate variance when natural-origin spawners are present (McElhany and Payne in prep; McClure et al. 2003). Conceptually, the correction involves modifying the N_{t+1}/N_t ratio (Table E.1).

Harvest can also mask a population's underlying variability, but we can apply corrections similar to those made for hatcheries (Table E.1) (McElhany and Payne in prep.). Although other human activities can potentially impact variability estimates, we apply the corrections to hatcheries and harvest primarily because we have *a priori* reasons to expect them to modify the variance and because data are available. Hatchery production has varied widely in some systems, leading us to suspect it influences variance estimates of the available time series. Most harvest strategies have the goal—explicit or implicit—of reducing variability on the spawning grounds. Thus we suspect that uncorrected variance estimates tend to underestimate natural variability. Applying the corrections in age-structured salmon populations requires estimating the average age of spawner return. An important issue regarding these corrections in practice is that we seldom know the measurement error in estimates of fraction of hatchery-origin spawners and the

Table E.1 Modifications to the N_{t+1}/N_t ratio to correct for harvest and hatchery impacts on the time series.

Correct for Harvest	Correct for Hatchery	Ratio ^a
No	No	$\frac{S_{t+1}}{S_t}$
No	Yes	$\frac{W_{t+1}}{W_t + \delta H_t}$
Yes	No	$\frac{S_{t+1} + C_{t+1}}{S_t}$
Yes	Yes	$\frac{W_{t+1} + C_{t+1}}{W_t + \delta H_t}$

^a These equations ignore the complications of age structure, which are dealt with in McElhany and Payne (in prep) and McClure et al. (2003).

Key:

S_t = total number of spawners

W_t = number of natural-origin spawners at time t

C_t = additional number of natural-origin fish that would have returned to spawn had there been no harvest

H_t = number of hatchery-origin fish that spawn in the wild

δ = reproductive success of hatchery-origin fish spawning in the wild relative to natural-origin fish

number of additional natural-origin fish that would have returned had there been no harvest. This uncertainty about the corrections input parameters may render the uncorrected estimates more reliable, even if hatcheries and harvests both influence the spawner time series.

Population-Specific Versus Pooled Variance Estimates

Because of differences in environmental conditions, every population probably has a different mean environmental variance. If we had precise and accurate estimates of the variances, we could use these data to parameterize population-specific viability curves. However, there is often much uncertainty surrounding the variance estimate, thus more accurate viability curves may be generated by pooling variance estimates from several populations, which can be averaged to produce a “generic” viability curve that can be applied to a number of populations. The PCC targets would likely still be different for all populations because target size is a function of current size, and populations likely differ in current abundance. If populations are pooled, the assumption is that they all have a similar environmental variance and that most observed differences in individual variance result from estimation error about a common mean; further, that the differences do not reflect the true underlying population-specific difference. We suspect that, in general, differences in variance estimates do not reflect population-specific

estimates because there is such a high level of uncertainty about any particular population estimate. For example, if the slope method is applied to 20 years of data, only three degrees of freedom are available for the variance estimate. This results in a high level of uncertainty about the true value of σ^2 (Figure E.1); if the point estimate is 0.05, there is roughly a 32% chance the true variance is greater than 0.1, which has a large impact on the viability curve. If populations are pooled, and it is assumed that every population represents an independent variance estimate, the point estimate becomes the average of the population variance estimates, and the degrees of freedom is the sum of the degrees of freedom from each population estimate. If populations are pooled such that there are 20 degrees of freedom, the probability that a point estimate of 0.05 comes from a sample with a true value of 0.1 drops to about 3%. The individual populations are likely to reflect independent measures of variability because populations are defined based on a high level of demographic independence. The approach we have taken thus far is to pool the estimates within an ESU to estimate environmental variance.

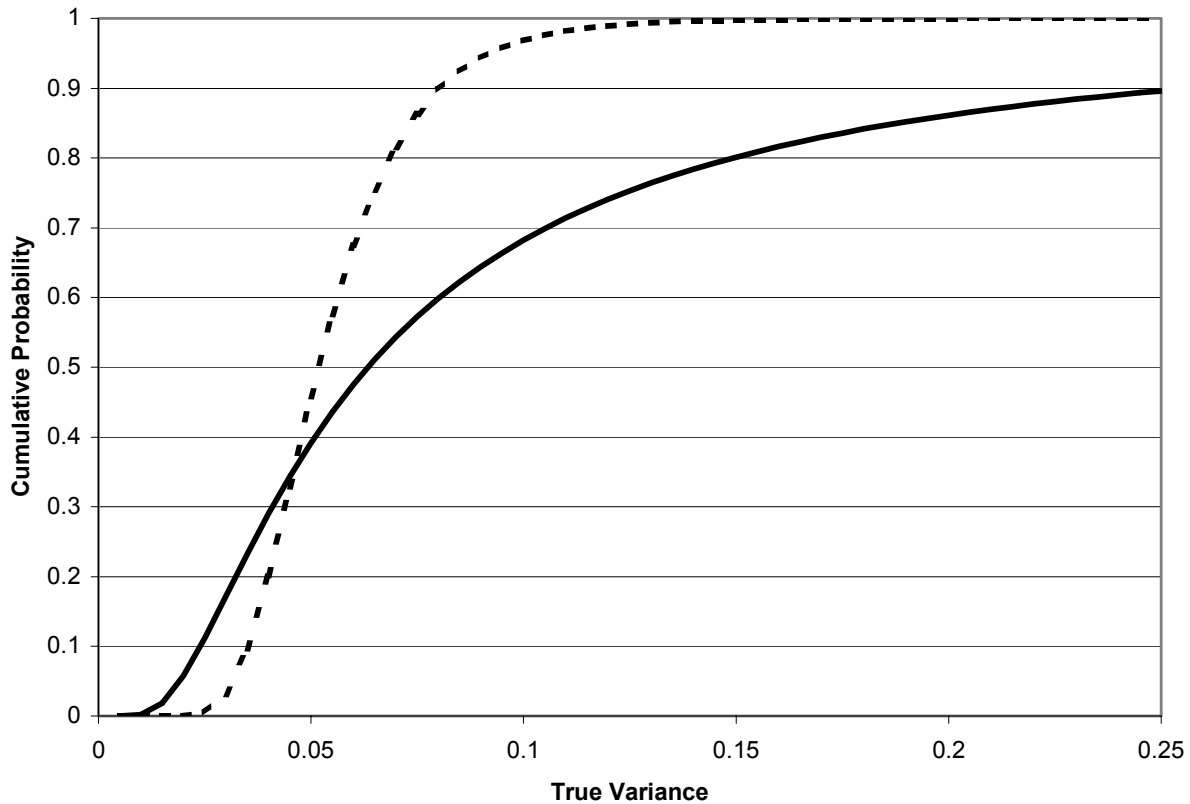


Figure E.1 Sampling distribution of variance estimate with a point estimate of 0.05. The solid line shows the distribution with 3 degrees of freedom, and the dashed line with 20 degrees of freedom.

Table E.2 Variance estimate diagnostics.

Assumption	Test
$\sigma^2 > 0$	Examine output of slope estimate
The relationship of τ is linear	R^2 of least squared fit of variance estimate versus τ .
τ is normally distributed	Test for significant outliers using dffits statistics > 2
No temporal trends in τ	Parametric significance test and R^2 of least squared fit of variance estimate versus time.
No serial autocorrelation in τ	Test still in development
No density dependence in time series	We have not yet conducted tests of density dependence. (These tests tend to have little power.)

Variance Estimate Diagnostics

Several diagnostic tests are available to evaluate whether the data in a time series are consistent with the assumptions of the basic demographic model. We explored tests that are similar (but not identical) to those of McClure et al. (2003) (Table E.2).

Variance Estimates for WLC Populations

The primary data needed to calculate the variance are time series of population spawner counts or of an index that is proportional to the population spawner counts. To apply the hatchery and harvest corrections we also need estimates of the fraction of hatchery-origin spawners present each year, the relative reproductive success of the hatchery-origin spawners, the number of additional natural-origin fish that would have returned had there been no harvest each year, and an estimate of the average age at spawning. We have obtained as many relevant time series as possible for populations in the Willamette/Lower Columbia domain. These time series, their references, and dataset descriptions are available on the Web at http://research.nwfsc.noaa.gov/cbd/trt/wlc_trt/viability_report.htm. A computer program that calculates the variance estimates with user provide inputs (including options for the harvest and hatchery corrections), SimSalmon version 4.5.3 beta, is available at the same Web site.

In estimating variance for WLC populations, we were limited to a large extent by available data. We explored the variance estimates under a number of different assumption options (Table E.3). The input data were collected using a variety of methods and are of mixed quality. The variance estimates and diagnostic outputs for the WLC populations under one set of options are shown in Table E.4. The variance estimates and diagnostics for all populations under all assumption option sets are available on the Web at http://research.nwfsc.noaa.gov/cbd/trt/wlc_trt/viability_report.htm. The average variance estimate by ESU and life-

Table E.3 Analysis options for estimating environmental variance from available time series in the WLC.^a

Option Number	Relative Fitness of Hatchery Origin Spawners^b	Includes Correction for Harvest?^c	Years Used for Analysis^d
1	0	No	All data
2	0.5	No	All data
3	1	No	All data
4	0	Yes	All data
5	0.5	Yes	All data
6	1	Yes	All data
7	0	No	Since 1980
8	0.5	No	Since 1980
9	1	No	Since 1980
10	0	Yes	Since 1980
11	0.5	Yes	Since 1980
12	1	Yes	Since 1980

^a The hatchery correction was applied to all options. Because of limited data availability or the history of the population, not all populations could be analyzed under all the options.

^b The relative reproductive success of hatchery-origin spawners compared to natural origin spawners assumed for a particular option.

^c Indicates whether or not the harvest correction was applied for a particular option.

^d Indicates whether or not the analysis used all available data or only data since 1980 for a particular option.

history type for each option is shown in Table E.5. Table E.6 shows the summed degrees of freedom associated with the averages in Table E.5. For a given ESU/life-history type, the variance averages are relatively similar under all assumption option sets.

Table E.4 Variance estimates and diagnostics for WLC populations, assuming that hatchery fish have the same reproductive success as natural-origin fish and with no harvest correction.^a

ESU	Population	Years of Data	Sample Size for (N_{t+1}/N_t)	Variance (95% Confidence Interval)	Variance Degrees of Freedom	Slope of (N_{t+1}/N_t) vs. Time	Number of Outliers from Normal Distribution
Lower Columbia chinook spring	Cowlitz River	1980–1999	16	0.015 (0.005–0.164)	3.37	n.s. ^b	1
Lower Columbia chinook salmon late fall	Lewis River	1964–2000	17	0.038 (0.017–0.152)	7.37	n.s.	1
	Sandy River	1984–2001	9	0.04 (0.013–0.591)	2.9	-0.043	0
Lower Columbia chinook fall	Big White Salmon River	1964–2000	17	0.175 (0.078–0.691)	7.37	n.s.	1
	Coweeman River	1964–2000	17	0.186 (0.083–0.735)	7.37	n.s.	1
	Cowlitz River	1964–2000	17	0.714 (0.317–2.817)	7.37	n.s.	0
	East Fork Lewis River	1980–2000	17	0.01 (0.003–0.094)	3.61	n.s.	0
	Elochoman River	1964–2000	17	0.381 (0.169–1.505)	7.37	n.s.	1
	Grays River	1964–2000	17	0.31 (0.138–1.224)	7.37	n.s.	1
	Kalama River	1964–2000	17	0.311 (0.138–1.226)	7.37	n.s.	1
	Mill Creek River	1980–2000	17	0.141 (0.049–1.382)	3.61	-0.028	0
	Washougal River	1964–2000	17	0.088 (0.039–0.346)	7.37	n.s.	0
	Wind River	1980–2000	12	0.361 (0.125–3.534)	3.61	n.s.	0
	Clackamas River	1967–2001	26	0.091 (0.04–0.384)	6.9	n.s.	2
Lower Columbia steelhead winter	Clackamas River	1958–2001	40	0.097 (0.046–0.321)	9.02	n.s.	2
	Kalama River	1977–2002	22	0.031 (0.012–0.197)	4.78	n.s.	2
	North Fork Toutle River	1989–2002	10	0.001 (0–0.053)	1.97	n.s.	1
	South Fork Toutle River	1984–2002	10	0 (0–0.002)	3.14	n.s.	2
	Sandy River	1978–2001	16	0.027 (0.01–0.202)	4.31	n.s.	0
	East Fork Lewis River	1985–1994	6	0.004 (0.001–3.798)	1.02	n.s.	0
	Hood River	1992–2000	5	0.041 (no estimate)	0.79	0.133	1

Appendix E: Environmental Variance in PCC Analysis

Lower Columbia steelhead summer	Kalama River	1977–2003	23	0.178 (0.07–1.068)	5.02	n.s.	2
	Washougal River	1986–2003	14	0.07 (0.022–1.049)	2.9	0.03	2
	Wind River	1989–2003	11	0.006 (0.002–0.182)	2.2	n.s.	1
	Hood River	1992–2000	5	0.01 (no estimate)	0.79	0.104	0
Lower Columbia chum	Grays River	1967–2000	28	0.051 (0.022–0.222)	6.67	n.s.	1
	Hardy Creek	1957–2000	40	0.076 (0.036–0.253)	9.02	n.s.	2
	Lower gorge	1944–2000	53	0.08 (0.041–0.216)	12.07	n.s.	3
Upper Willamette chinook salmon spring	Clackamas River	1958–2002	41	0.107 (0.051–0.348)	9.25	n.s.	3
	McKenzie River	1970–2001	28	0.122 (0.051–0.572)	6.19	n.s.	1
Upper Willamette steelhead winter	Calapooia River	1980–1997	14	0.211 (0.067–3.147)	2.9	n.s.	1
	Molalla River	1980–1997	14	0.072 (0.023–1.068)	2.9	n.s.	0
	North Santiam River	1980–1997	14	0.066 (0.021–0.984)	2.9	n.s.	1
	South Santiam River	1980–1997	14	0.008 (0.002–0.113)	2.9	n.s.	0

^a Option 3 in Table E.3.

^b n.s. indicates that slope is not significant at $\alpha = 0.05$.

Table E.5 Average variance estimates for WLC ESU/life-history types under a number of different assumption sets.^a

Option Number	Lower Columbia					Chum	Upper Willamette	
	Chinook Salmon			Steelhead			Chinook	Steelhead
	Spring	Late Fall	Fall	Winter	Summer		Spring	Winter
1	0.015	0.039	0.251	0.030	0.063	0.069	0.114	0.095
2	0.015	0.039	0.261	0.029	0.065			0.089
3	0.015	0.039	0.252	0.029	0.066			0.089
4		0.081	0.288	0.027	0.069			
5		0.080	0.287	0.027	0.064			
6		0.080	0.287	0.028	0.064			
7	0.015	0.039	0.251	0.029	0.064			0.095
8	0.015	0.039	0.261	0.029	0.065			0.089
9	0.015	0.039	0.268	0.029	0.065			0.089
10		0.081	0.288	0.023	0.064			
11		0.080	0.287	0.022	0.062			
12		0.080	0.287	0.022	0.062			
Total Average	0.015	0.053	0.271	0.027	0.065	0.069	0.114	0.091

^a Table E.3 regarding assumption options. Because of data availability, some ESU/life-history types could not be evaluated under some assumption options.

Table E.6 Summed degrees of freedom estimates for WLC ESU/life history types under a number of different assumption sets.^a

Option Number	Lower Columbia					Chum	Upper Willamette	
	Chinook Salmon			Steelhead			Chinook	Steelhead
	Spring	Late Fall	Fall	Winter	Summer		Spring	Winter
1	3	10	62	25	11			12
2	3	10	62	25	11			12
3	3	10	69	25	11	28	15	12
4		7	59	23	10			
5		7	59	23	10			
6		7	59	23	10			
7	3	7	36	19	10			12
8	3	7	36	19	10			12
9	3	7	36	19	10			12
10		4	32	17	9			
11		4	32	17	9			
12		4	32	17	9			

^a See Table E.3 regarding assumption options. Because of data availability, some ESU/life-history types could not be evaluated under some assumption options. These are the summed degrees of freedom that accompany the variance averages in Table E.5.

Preliminary Variance Estimate Conclusion

The average variance estimates by ESU and life-history type ranged from 0.015 for Lower Columbia River spring chinook salmon to 0.287 for Lower Columbia fall chinook (Table E.5). The lowest single population variance estimate was the Wind River winter steelhead, at 0.006; the highest was Cowlitz fall chinook, at 0.714 (Table E.4). The average of the ESU/life-history averages is approximately 0.08. The Lower Columbia fall chinook had consistently higher variance estimates than other ESU/life-history types. This may reflect some inherently higher variability in the Lower Columbia fall chinook populations; alternatively, it may reflect high levels of measurement error in the abundance time series. The Lower Columbia fall chinook populations tend to have large fractions of hatchery-origin spawners, but the Washington Department of Fish and Wildlife (WDFW) considers actual hatchery fraction estimates to be very imprecise. It is interesting to note that the Lower Columbia population with little hatchery input (Coweeman) has a variance estimate of 0.187, which is below the average for Lower Columbia fall chinook but above the average for other ESU/life-history types.

Based on examination of the data and diagnostic output, we tentatively applied a variance estimate of 0.05 for all ESU populations in the WLC domain and assumed 20 degrees of freedom. This is not based a single mathematical calculation but on a professional judgment evaluation that incorporated the estimated average variances and assessment of overall data quality of individual time series. This assessment led to a discounting of the variance estimates from the Lower Columbia fall chinook population for reasons discussed in the previous paragraph. The variance estimate of 0.05 is just an initial starting point; its accuracy, used in conjunction with the PCC targets, would be expected to improve with additional high-quality time-series data.

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APPENDIX F

CONCERNS RAISED ABOUT POPULATION CHANGE CRITERIA APPROACH

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In reviews of a previous draft of this document a number of concerns were raised about the population change criteria (PCC) approach. A summary of some of these concerns and brief responses follow. More detailed discussion of these issues is found in Appendix D.

Concern Raised: **PVA models are not predictive enough to set thresholds.**

Population viability analysis models (PVAs) attempt to predict low-probability events for long time spans into the future. This can be notoriously imprecise. Fieberg and Ellner (2000) suggest that it is possible to predict into the future only 1/10 the length of the time series of available data. Reed et al. (2002) recommend against the use of PVAs to set minimum population sizes. We agree completely that PVA models, such as the one we use, should be interpreted with extreme caution, particularly with regard to minimum sizes. However, these models tend to be most sensitive to error when the intrinsic productivity is near the threshold of 1. If a population clearly has an intrinsic productivity less than 1, the models robustly predict that a population will go extinct. Conversely, if the intrinsic productivity is clearly greater than 1, the models robustly predict that a population will persist. The conclusion is robust in the sense that it is not overly sensitive to estimates of variability or to the actual abundance of the population. If a productivity estimate is near 0, the extinction prediction is highly sensitive to the variance and abundance estimate. (These points are illustrated in Fieberg and Ellner 2000, Figure 1.) The PCC approach focuses on estimating the population's productivity and demonstrating that it is greater than 1. This takes advantage of the situation in which PVA models provide the most predictive power.

Concern Raised: **Approach is nontraditional and ignores common fisheries modeling to estimate productivity.**

The basic question addressed by traditional fisheries modeling is different than the question addressed in setting viability criteria. Traditional fisheries models are designed to address the question, "How much of this healthy population can be harvested?" The question posed by delisting criteria is, "What is the best way to determine when a threatened population is

no longer in danger of extinction?” Although different in some key respects than traditional fisheries models, the PCC approach shares many features. Like traditional fisheries modeling, it considers intrinsic productivity, density dependence, and variation to predict future population performance. The main difference is in the method used to estimate productivity. The PCC approach is not expected to provide a measure of the intrinsic productivity, but rather to provide a statistically defensible bound on the productivity. Reviewers of this approach have suggested that fitting recruitment curves to recruits per spawner data is the way to determine the “true” intrinsic productivity. As demonstrated in Appendix D, recruits per spawner data often provide little information on the true intrinsic productivity, and alternative methods are needed. In those exceptional cases in which recruits per spawner data are informative, we recommend evaluating population status using those data.

Concern Raised:

Approach sets different abundance levels for different populations, not a fixed number.

The extinction risk to a population depends on both a population’s productivity and its abundance. A fixed minimum abundance threshold for all populations would only be appropriate if all populations had the same productivity. One approach to setting a fixed minimum size might be to assume that all populations had a productivity of 1. The minimum sizes suggested by this assumption tend to be quite large, and this is also the range over which PVA models are most uncertain. An alternative approach might be to assume (set as a target) a productivity greater than 1. If the productivity is set, for example, at 1.1, under the PVA models evaluated the minimum population size for a low extinction risk drops to a range of a few hundred fish. The critical issue becomes not setting and evaluating a minimum abundance threshold, but evaluating the population’s productivity. The PCC approach evaluates productivity by measuring population growth. The population growth rate is estimated based on the difference between the current abundance and a target future abundance; different populations will have different targets because they have different current abundances. The PCC approach actually does not use a fixed productivity threshold; rather, it simultaneously examines population abundance and productivity. As a consequence, populations that are currently small must show a greater proportional increase in abundance than populations that are currently large.

Concern Raised:

**Approach requires all populations to increase from current abundance, even those that are relatively large and stable.
(Or, “What about the Lewis River brights?”)**

The population change approach relies on observed growth rate as an estimate of intrinsic productivity. The approach is most appropriate when applied to populations that have been depressed below historical abundance. If a population is relatively large and apparently stable, an intuitive conclusion is that the population has a low risk of extinction. However, the intuitive

perception depends explicitly or implicitly on an assumption that the population has some resilience (i.e., intrinsic productivity > 1), because even quite large populations can have a substantial risk of extinction if they have no resilience. The challenge once again is to demonstrate that a population has an acceptable intrinsic productivity. Different null hypotheses regarding the growth rate may be appropriate depending on the situation. For example, if we observe a pristine population fluctuating around historical abundance, we would not need to observe the population grow to conclude that it is sufficiently resilient to persist. The assumption of historical condition would be enough evidence to reach that conclusion. However, if we observe a population depressed to a small fraction of its historical abundance, we may require substantial statistical evidence before concluding that a population has an adequate intrinsic productivity. In limited cases, the statistical evidence may be provided by fitting recruitment curves to observed recruits per spawner data. In these cases, adequate resilience may be concluded without observing an actual population increase. However, in the majority of cases, recruits per spawner data are uninformative regarding intrinsic productivity, and the PCC are a useful method of providing the needed statistical rigor. It is important to note with regard to the Lewis River bright chinook salmon population that the target for a category 3 population is actually lower than the average abundance over the last 20 years, which suggests that the criteria are not unattainable, even for this relatively large population.

Concern Raised:
The model is sensitive to the begin and end dates
for the growth rate estimates.

The median annual growth rate is conceptually based on a formula that includes data from every year:

$$\hat{\lambda} = \exp \left(\text{mean} \left(\ln \left(\frac{N_{t+1}}{N_t} \right) \right) \right).$$

However, this equation simplifies to

$$\hat{\lambda} = \exp \left(\frac{1}{y} \ln \left(\frac{N_{\text{target}}}{N_{\text{initial}}} \right) \right),$$

where y is the number of years between the initial abundance and target abundance counts. Thus, estimating the median annual growth rate is a function of the initial and target population sizes and is sensitive to the dates selected for these periods. Three features of the PCC as they have been developed help reduce the sensitivity to the selected start period date. First is the use of a four-year average for the abundance estimates, which tends to smooth out much of the interannual variation. The second feature is requiring a relatively long observation period, which increases the likelihood of picking up the true underlying growth rate. The third feature is the marine survival rate correction, which attempts to correct for the marine regime shifts. It is

important to note that the PCC are intended to provide initial targets, and that we expect biologists in the future, when evolutionarily significant units (ESUs) are actually being contemplated for delisting, to perform quantitative risk analysis with the tools available at that time and to explore the consequences of the time frames evaluated.

Concern Raised: Model is too sensitive to QET.

Population viability models tend to be sensitive to the quasi-extinction threshold (QET) value. The QET value is the abundance below which the population should not drop, either because of increased extinction risk or uncertainty. Setting the QET value is difficult and somewhat arbitrary (e.g., there is no real scientific way to distinguish between the appropriateness of a QET of 50 spawners or 60 spawners.) The PCC approach is moderately sensitive to QET. Because it depends on estimates of intrinsic productivity greater than 1, the results are much less sensitive than minimum size estimates, assuming that intrinsic productivity equals 1. For details on the reasons for this conclusion, see Appendix D.

Concern Raised: Stationarity assumption about variance.

In setting targets in this report, we assume that the environmental variance observed for the recent past is predictive of the environmental variance we will observe in the future. We recognize that this parameter may change in response to management actions, and we encourage the constant reassessment of this parameter. However, we note that detecting changes in environmental variance is extremely challenging and requires long time series of abundance.

Concern Raised: Population change criteria need to be met once.

This concern actually raises several issues, one of which is regarding the stationarity assumption. The stationarity assumption is that a population's behavior over the observation period will continue into the future. This is a basic assumption, and one that confronts any effort to predict the future based on data collected during an observation period. For example, if intrinsic productivity were estimated by fitting a recruitment curve to recruits per spawner data, the intrinsic productivity estimate would constitute a criterion that is met once. In order to reach a conclusion about a population's risk status, a stationarity assumption needs to be applied. A second part of this concern involves issues about annual variability and the possibility of meeting the criteria by chance. The approach used to set the target abundances explicitly considers annual variability, and the extinction risk associated with the target considers the uncertainty surrounding the population growth rate estimate. The sensitivity to single-year variation is addressed partially by evaluating four-year averages, not single years.

Confusion Expressed: When is the status of the population evaluated?

The target growth rates in Table 4.2 of the main text are for a 20-year observation period. To evaluate the status of a population relative to these criteria, it would be necessary to compare the size of the population in 20 years to the target size. The population would not automatically be considered viable *even if it exceeded the target abundance at some point prior to the end of the 20 years*. This is because the PCC consider the length of the observation period as an import parameter in estimating the target. Target abundances can be easily calculated for shorter or longer observation periods, but the targets will likely differ from those in Table 4.2 if the observation period is other than 20 years (targets for different observation periods are provided in Appendix D). The relationship between target size and observation period involves tradeoffs between two factors. If the same abundance is reached in a shorter time, it implies a higher growth rate and a decreased probability that the population will go extinct. However, a shorter observation period leads to increased uncertainty, which tends to increase our estimate of the probability that the population may go extinct. The exact balance between these opposing tendencies can only be determined by doing the calculation. Theoretically, a target abundance could be calculated for every year into the future and compared to the observed abundance. This approach has some merit; however, a minimum number of observation periods are required to obtain any precision with growth rate estimates. Work by Holmes (2002) suggests that a minimum of 12 to 15 years of data are needed, assuming there are no long-period (decadal-scale) cycles or regime shifts in marine survival. Given that there are long-period cycles or regime shifts in marine survival, the observation period should be as long as possible to average over as much of the range of marine survivals as possible. For this reason, the Willamette/Lower Columbia Technical Recovery Team (WLC-TRT) suggests an observation period of about 20 years. The target abundances based on a 20-year observation period shown in Table 4.2 are intended as general guideposts for population risk criteria. If the criteria were set as a target over 20 years, it would NOT be necessary to wait 20 years to evaluate whether the population is headed in the right direction. It would be possible to estimate the likelihood that a population is on track to make the goal from a shorter time series. Such an estimate would be imprecise (too imprecise to conclude viability), but would indicate whether the population is improving or declining.

Concern Raised: The targets cannot be completely predetermined.

The targets should be viewed as initial estimates of the target abundances, not as the final answer carved in stone. On a general note, there will undoubtedly be advances (or at least modifications) in risk assessment methods over the next several decades, and we expect criteria to be regularly reevaluated and modified. Considering the approach we have developed, the criteria cannot be completely specified in advance because the abundance target, which is conditional on several key parameters, must be examined retrospectively to determine risk. To

evaluate a population's viability using the PCC approach requires estimating the fraction of hatchery-origin spawners that effectively spawn in the wild and marine survival during the observation period. Managers could theoretically predict the number of hatchery-origin spawners in the wild because that is in large part under human control. However, the WLC-TRT has not been provided with any projections, so information on this parameter could not be incorporated into the targets. Even if projections were provided, they would be just that—projections; it would be necessary to wait and observe the actual pattern of hatchery spawning. Since the level of hatchery spawning in the wild is expected to change over time, providing targets in advance is especially difficult. The marine survival parameter also cannot be predicted with precision; it must be evaluated retrospectively to determine whether the population has reached sufficient size over the observation period.

Concern Raised:
Model does not provide guidance on actions.

The PCC approach is one tool for evaluating whether a threatened population is still in danger of extinction. It is not intended to provide guidance on what actions should be taken to recover populations. It is intended to evaluate whether the cumulative effect of all actions has accomplished the objective of reducing the risk of extinction.

Concern Raised:
Low abundance default is arbitrary.

Many populations in the WLC domain are extirpated or currently at very low abundance. Because the PCC have an increased uncertainty at very small population sizes, and cannot be calculated at all for an extirpated population, a low-abundance default was applied. The low-abundance default is an assumption about the current population size. The larger the assumed current population size, the higher the target needed to reach a given persistence probability. The selection of the low-abundance default value is based on professional judgment and is informed by an understanding of the processes that contribute to uncertainty at small population sizes. However, there is no quantitative justification of the value selected. In practice, it may be advisable to wait and develop a population change target after the population has increased in abundance sufficiently to obtain a relatively precise estimate of the population size.

Concern Raised:
Targets are not established for all populations.

Developing PCC targets requires an estimate of the current spawner abundance. For some populations in the WLC domain, adequate data were not available to estimate current population abundance. Before targets can be developed for these populations it is necessary to obtain an estimate of the four-year average abundance. It is not appropriate to apply the low-abundance

default to these populations because if the actual abundance is greater than the default, the population change targets would be underestimated.

**Concern Raised:
Single variance estimate for all populations.**

Every population is unique and likely has its own pattern of response to environmental variation. However, estimates of variation from individual populations are very uncertain, and a better estimate can often be obtained by averaging the estimates from multiple populations. As variance estimates are refined by the collection of more data over time, it is hoped that population—or at least ESU-specific estimates—can be developed.

**Concern Raised:
The approach is too complex.**

The PCC are not as easy to explain as a simple abundance threshold. However, the basic concepts underlying them are relatively simple. A viable population must be resilient. A reasonable way (but not the only way) to estimate resilience is by observing a population's growth rate, which is measured by a change in abundance over time. The PCC work out in advance how much change is required over a given time to conclude the population has a low probability of extinction. Although understanding the mathematics and statistics underlying the calculations may require specialized expertise, explaining the basic results and consequences of the criteria to watershed planners should not. The basic message is that populations need to increase in abundance, and Appendix D gives some ballpark indication of how much and how fast. It is true that the criteria are not as easy to explain as a simple abundance threshold. However, the criteria address the key issue of productivity in addition to abundance, and the slight additional complexity is worth the extra effort in explanation. Any criteria approach that requires measuring productivity is going to be inherently more complex than a simple abundance threshold. For example, the alternative recruits-per-spawner approach might be familiar to fisheries biologists, but it is not a trivial thing to explain to a broader audience.

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APPENDIX G

EXPLORATION OF RECRUITS PER SPAWNER ANALYSIS

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Overview

All models are wrong, some are useful.
—Box 1976

A generic approach for identifying a viable productivity-abundance criterion involves estimating extinction risk using a population dynamics model and determining the threshold where productivity and abundance parameters just yield an acceptable risk. The results of this sort of analysis can be plotted in a “viability curve,” where every point on the curve represents a productivity-abundance combination with identical extinction risk (Figure G.1). A key issue in developing a specific method from this generic approach is defining the form of the population dynamics model used to estimate extinction risk. In Table G.1, we describe a number of relatively simple population dynamics models that have been applied to salmon and could potentially be used to estimate extinction risk. Many of these models are discussed in Hilborn and Walters (1992).

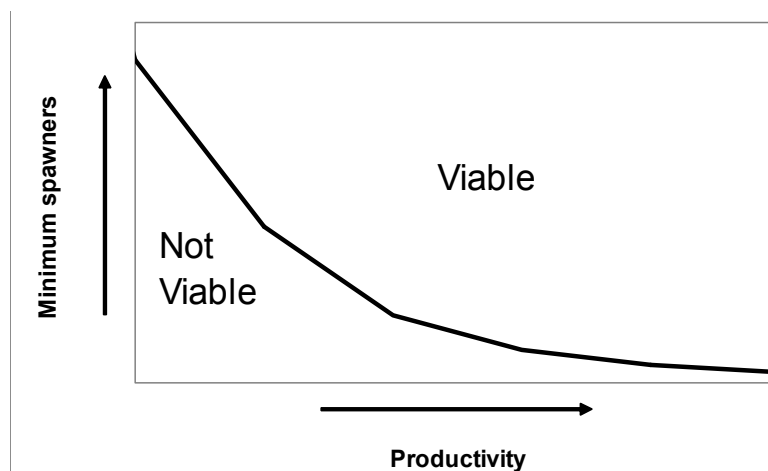


Figure G.1 Conceptual graph of the relationship between productivity, population size, and extinction risk. The curve in the figure represents combinations of size and productivity that just exactly have the acceptable extinction risk.

All of these models except the “constant recruitment” model involve projecting the number of recruits as a stochastic function of the number of spawners. The constant recruitment model assumes that recruitment fluctuates stochastically around some fixed value, regardless of the number of spawners. In addition to a parameter describing the variation in the spawner-recruit relationship, some of the models include additional terms, such as carrying capacity or marine survival. It is important to emphasize that recruitment curves describe an average relationship between recruits and spawners from which individual years will surely deviate, and to reiterate the point made in the opening quote of this appendix, none of the models in Table G.1 describes the true relationship between recruits and spawners. The challenge is determining whether any of them may be useful for setting viability criteria. We return to this point in the section below on model selection.

Table G.1 Population dynamics models proposed for salmon populations.

Model Number	Model Name	Equation ^a
Model 0	Random walk	$R = S \exp(\sigma_0 Z)$
Model 1	Random walk with drift; stochastic exponential growth or decline	$R = S \exp(a_1 + \sigma_1 Z)$
Model 2	Constant recruitment	$R = b_2 \exp(\sigma_2 Z)$
Model 3	Stochastic hockey stick; stochastic exponential growth with a ceiling	$R = \min(S, b_3) \exp(a_3 + \sigma_3 Z)$
Model 4	Ricker; stochastic logistic	$R = S \exp(a_4 + b_4 S + \sigma_4 Z)$
Model 5	Beverton-Holt	$R = \frac{a_5 S}{1 + \frac{a_5}{b_5} S} \exp(\sigma_5 Z)$
Model 6	Ricker juvenile production with given marine survival	$R = c_6 S \exp(a_6 + b_6 S + \sigma_6 Z)$

^a In the equations,

S_t = the number of spawners

R = the number of recruits

Z = a unit normal random variable

$\sigma_{\#}$ = the standard deviation of the process error

$a_{\#}$ and $b_{\#}$ = equation-specific parameters, with the $a_{\#}$ parameter relating in some way to “intrinsic productivity” and the $b_{\#}$ parameter relating in some way to “capacity”

c_6 = a marine survival parameter; the a_6 and b_6 parameters in this equation relate to the production of juvenile outmigrants from spawners

Viability Curves

In Figures G.2 to G.5, we present several viability curves associated with the recruitment functions in Table G.1. The extinction risk associated with any particular parameter combination for a given model is found by simulating a large number of population trajectories and counting the fraction of trajectories that drop below the quasi-extinction risk threshold within the given time horizon. The intrinsic productivity axis in the curves refers to the number of recruits per spawner at very low (approaching 0) abundance. Exactly how the intrinsic productivity value relates to extinction risk depends on the specific form of the population dynamics model. In all the models, the intrinsic productivity provides an indication of population resilience, which is the tendency of the population to return toward an equilibrium value if perturbed to low abundance. The abundance axis in the curves refers to the point estimate equilibrium abundance. The initial population size for the population trajectories was the equilibrium (or mean equilibrium) abundance value for the Beverton-Holt and Ricker curves, and the carrying capacity for the hockey-stick curve. The shape of the viability curve was found by a grid search of the parameter space to identify productivity-abundance combinations with equivalent risk. This meant varying

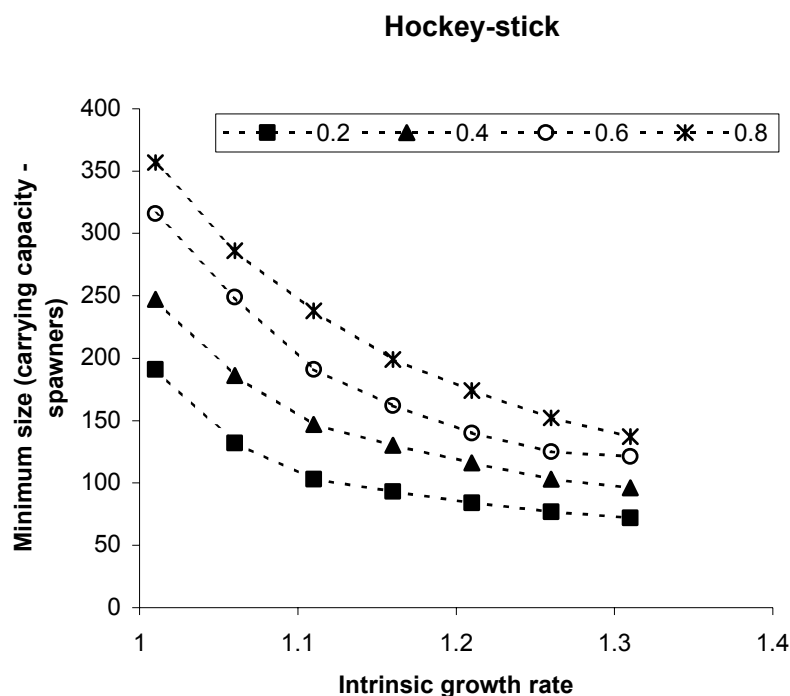


Figure G.2 Viability curves based on hockey-stick recruitment function. The different curves are for different levels of environmental variability. The viability curves were generated for a semelparous population where the average percentages of individuals spawning at a given age are: age 1 = 0%, age 2 = 1%, age 3 = 19%, age 4 = 57%, and age 5 = 23%. This life-history structure is typical of that observed for chinook salmon. In this model, the equilibrium abundance is the carrying capacity. Every point on a curve has the same extinction probability. In this example, the extinction probability is a 5% probability of declining to a four-year average of 50 spawners in 100 years.

the productivity, capacity, and process error variance parameters (i.e., the a , b and σ^2 parameters in Table G.1) and fixing all other parameters. In addition to the equilibrium abundance, the figures show the viability curves in terms of the “carrying capacity.” The carrying capacity has different biological interpretations for the different models, so they are not directly comparable. However, the shape of these capacity curves is informative.

A common feature of all the viability curves we have examined is that as the intrinsic productivity parameter exceeds about 1.1, the number of spawners needed for a viable population (i.e., a population that has a risk of less than 5% of declining to a four-year average of 50 fish in 100 years) declines to a few hundred fish. For the example, Ricker and Beverton-Holt curves in Figures G.3 and G.4, the viable equilibrium abundance is less than 200 spawners and relatively constant as long as the intrinsic productivity parameter is above 1. The parameter that varies more substantially in these models is the carrying capacity parameter, although it is a parameter we can never directly observe. If a population can be demonstrated to have an intrinsic productivity substantially above 1, the actual abundance of the population becomes much less relevant. A resilient population will likely be viable, even if it is very small.

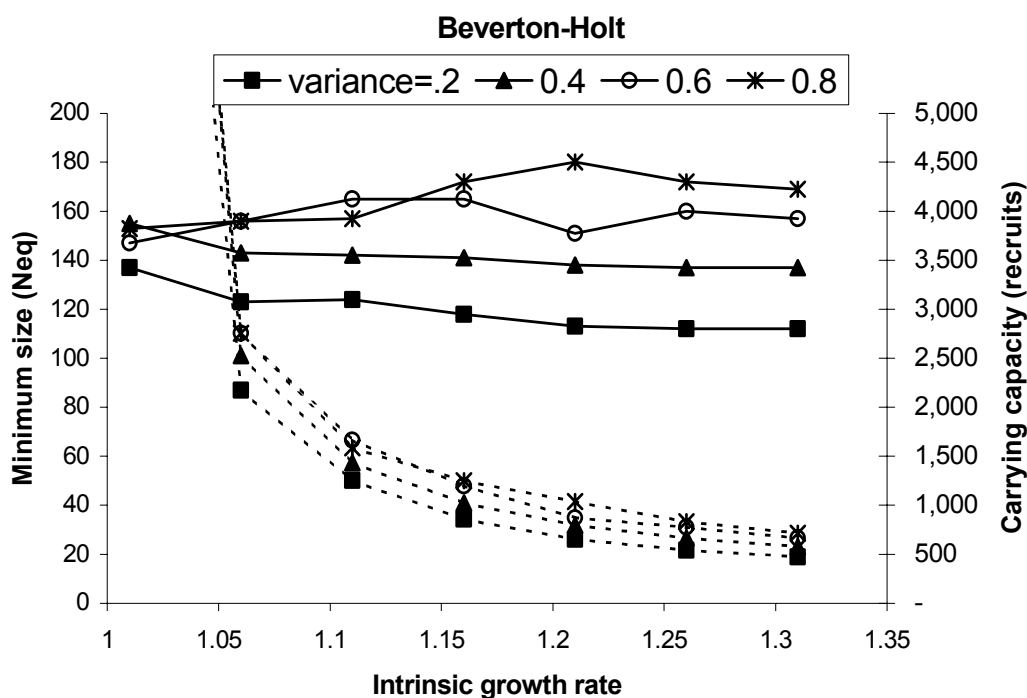


Figure G.3 Viability curves based on Beverton-Holt recruitment function. The different curves are for different levels of environmental variability. The viability curves were generated for a semelparous population where the average percentages of individuals spawning at a given age are: age 1 = 0%, age 2 = 1%, age 3 = 19%, age 4 = 57%, and age 5 = 23%. This life-history structure is typical of that observed for chinook salmon. The solid lines show equilibrium abundance and the dashed lines show the value of the “capacity” parameter in the Beverton-Holt function. Every point on a curve has the same extinction probability. In this example, the extinction probability is a 5% probability of declining to a four-year average of 50 spawners in 100 years.

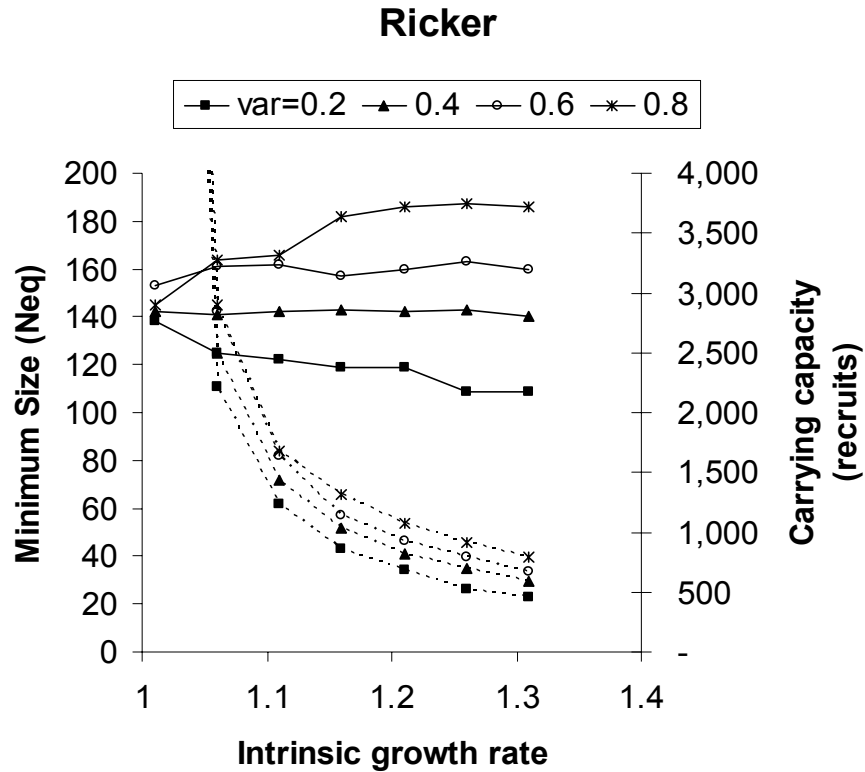


Figure G.4 Viability curves based on Ricker recruitment function. The different curves are for different levels of environmental variability. The viability curves were generated for a semelparous population where the average percentages of individuals spawning at a given age are: age 1 = 0%, age 2 = 1%, age 3 = 19%, age 4 = 57%, and age 5 = 23%. This life-history structure is typical of that observed for chinook salmon. The solid lines show equilibrium abundance, and the dashed lines show the value of the “capacity” parameter in the Ricker function. Every point on a curve has the same extinction probability. In this example, the extinction probability is a 5% probability of declining to a four-year average of 50 spawners in 100 years.

Ricker JOM with Cyclic Marine Survival

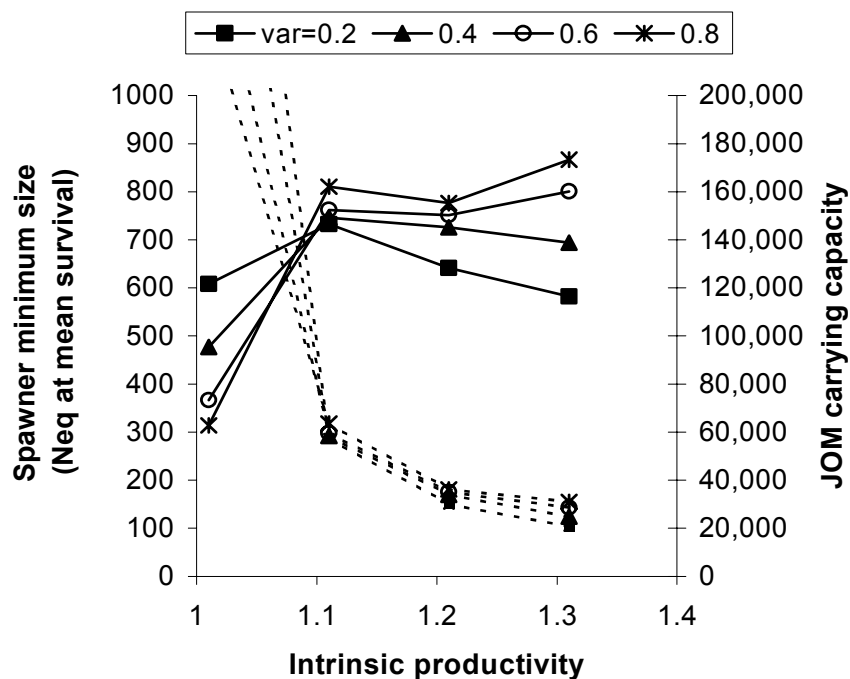


Figure G.5 Viability curves based on Ricker recruitment function for juvenile outmigrants with a cycle in marine survival. The different curves are for different levels of environmental variability. The viability curves were generated for a semelparous population where the average percentages of individuals spawning at a given age are: age 1 = 0%, age 2 = 1%, age 3 = 19%, age 4 = 57%, and age 5 = 23%. This life-history structure is typical of that observed for chinook salmon. The intrinsic productivity refers to the production of juvenile outmigrants. The ocean cycle in survival was a sine wave of 40 years' length with a mean survival of 0.05 and an amplitude of 0.03, beginning in year 0 of the cycle. The solid lines show equilibrium abundance of spawners, and the dashed lines show the value of the "capacity" parameter in the Ricker function for juveniles. Every point on a curve has the same extinction probability. In this example, the extinction probability is a 5% probability of declining to a four-year average of 50 spawners in 100 years.

Estimating Intrinsic Productivity

A key to evaluating a population's viability of using this approach is to estimate the intrinsic productivity. One of the great challenges with this general approach is determining which model, if any, might be appropriate for estimating intrinsic productivity. We can potentially look to existing abundance time series to determine which of the potential models is the "best approximating model" for this purpose (Burnham and Anderson 1998). Figure G.6 is an example of a spawner abundance time series. With information about the age structure of the population (and in some cases, numbers of hatchery spawners), it is possible to estimate how many recruits were naturally produced from each year's spawning (Figure G.7). To determine

which of the proposed model forms may be useful as approximating models for setting criteria, parameters for each model were estimated from available time series and the models were statistically compared (e.g., Figure G.8). Formal model selection analysis has been relatively rare in fisheries management, and models are often adopted without adequate consideration of the alternatives.

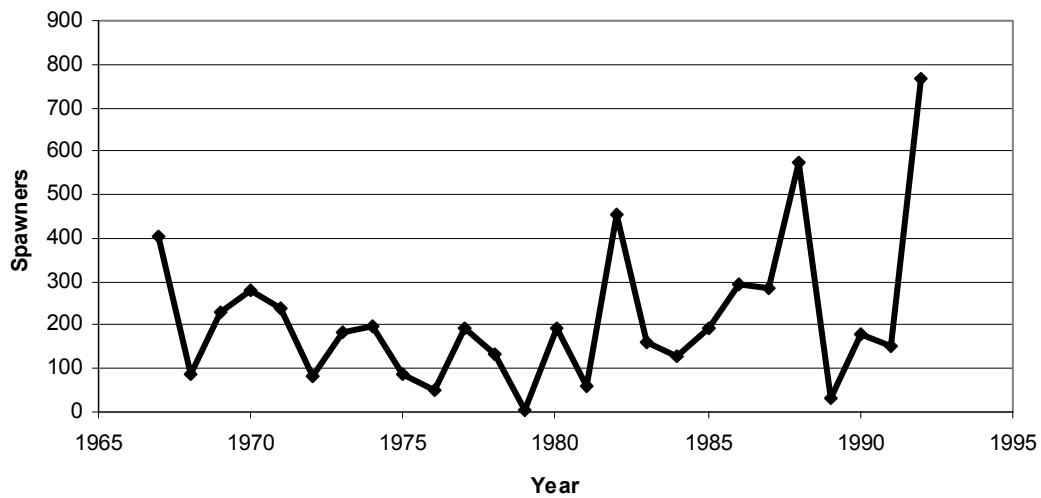


Figure G.6 Lower Columbia Gorge tributary chum salmon spawner abundance.

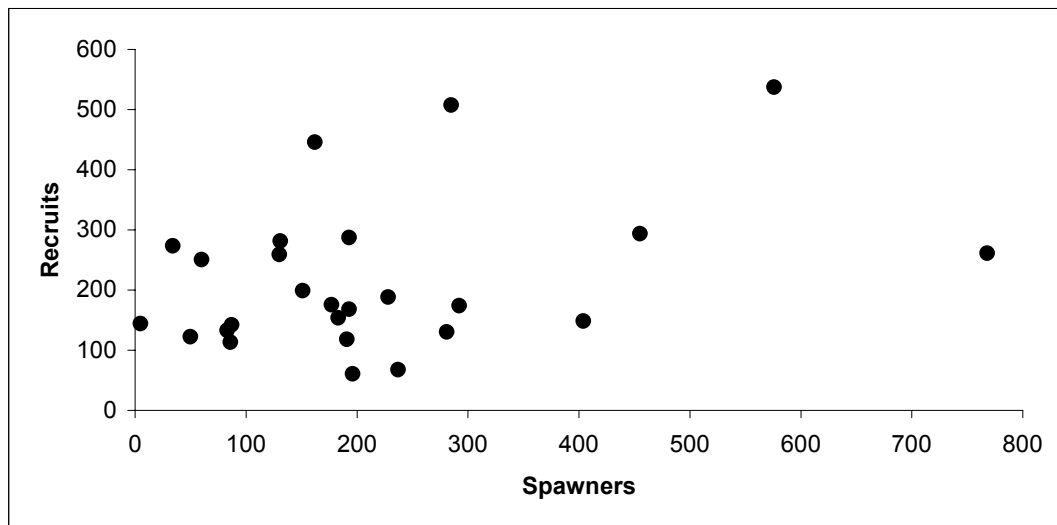


Figure G.7 Lower Columbia Gorge tributary chum salmon recruits versus spawners.

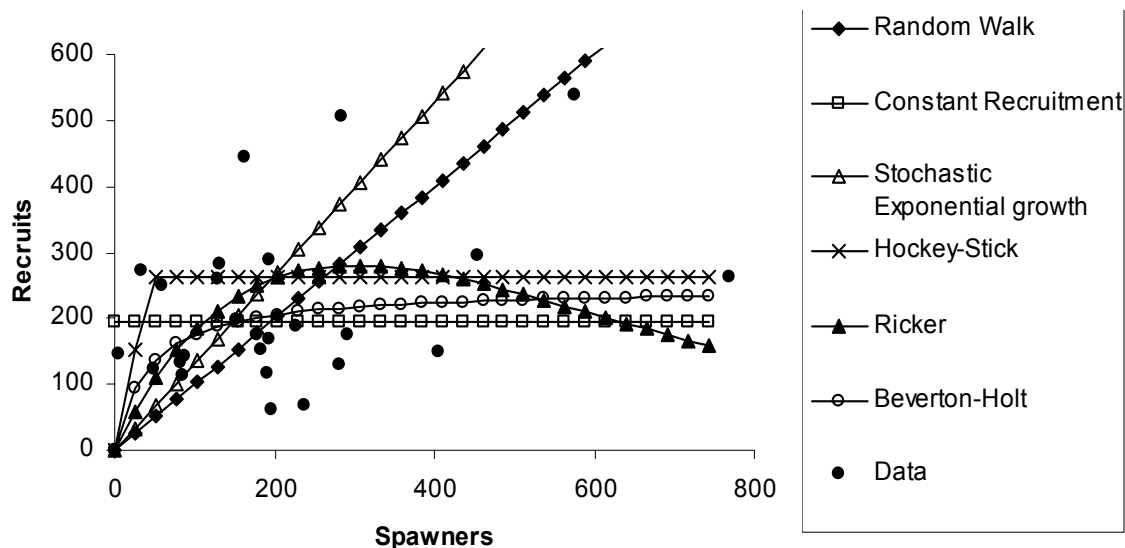


Figure G.8 Recruitment curves for lower Columbia Gorge tributary chum salmon.

Comparing Models

A number of different approaches exist for evaluating the relative utility of nonnested models. Among these approaches, the Aikake Information Criterion (AIC) method (Akaike 1973) addresses the question that is most relevant; i.e., how much do each of the models differ from the “true” process (Burnham and Anderson 1998). The AIC is a standard statistical measure of how well a model fits a data set, given a parameter set and assumptions about the error distribution. It is useful because it penalizes models in proportion to the number of parameters they have, without which we would not be able to compare larger models with smaller models. We used a version of the AIC that is corrected for small sample sizes: $AIC_c = 2(-\ln(L)) + (2p + 2 * p * (p + 1)) / (n - p - 1)$ (Burnham and Anderson 1998).

Other methods that can be used to select among models include likelihood ratio tests and comparing R^2 values associated with each model. In addition, several tests have been developed specifically to detect density dependence in abundance time series (e.g., Dennis and Taper 1994, Bulmer 1975, Pollard et al. 1987, Ruesink 2000, Shenk et al. 1998). All these methods are conceptually different from the AIC approach and have fundamental theoretical limitations. In our analysis, we have concentrated on the AIC evaluation.

We have estimated parameters for all available WLC spawner data sets for models 0–5 in Table G.1, then calculated AICc difference values to identify the best approximating models. The results are in Table G.2a-g. There are no fixed thresholds for interpreting the AIC difference values, but there are some general rules of thumb (Burnham and Anderson 1998). The AICc difference value for the best model is 0. If the AIC difference between the best model and the model with the lowest AIC is less than 2, then the second model provides a very good approximation relative to the best model, and the models might be given equal consideration. If the AIC difference value is greater than 10, the model is not a very good approximating model

relative to the best model and is not very well supported by the data. If the AIC difference is between 2 and 10, the interpretation is less clear, and the biology of the situation and the question being asked should drive how the model is considered.

For 20 of 32 populations examined, the best approximating model identified using the AIC method was the constant recruitment model (Figure G.9). The Ricker model was identified as the best approximating model for six populations, but for four of them the AIC difference value for the constant recruitment model was less than 2, and either the Beverton-Holt or the hockey-stick models were also within 2, so the models are approximately equally good. For 12 of the 32 populations, at least one of the density-dependent recruitment models (i.e., hockey-stick, Ricker, or Beverton-Holt) had a low AIC difference value and could be considered a contender as the best approximating model (Figure G.10). The constant recruitment model is interesting because it is the only model examined that assumes there is no relationship between the number of spawners and the number of recruits; the number of recruits is assumed to fluctuate around a constant value. The constant recruitment model is biologically implausible if extrapolated to very low spawner numbers because at the extreme, zero spawners must yield zero recruits. However, the fact that this model was selected as the best approximating model suggests that there is little data in the range of very low abundance to exclusively support one of the models that explicitly includes a relationship between spawners and recruits. Because there are few data at low abundance, there is very little information from which to estimate the intrinsic productivity. This is also reflected in the large confidence intervals on the intrinsic productivity estimates of individual models. The ability to estimate intrinsic productivity as evaluated by the model selection analysis does not seem to improve with increased length of the time series (Figures G.11 and G.12).

Table G.2.a Lower Columbia River late-fall (bright) chinook salmon population parameter estimates and model comparison. Except as noted, recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.) Exceptions to these spawner and recruit definitions occur because of data limitations.

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Lewis River	Random walk			0.86 (0.61-1.11)	37.50	10.82
	Constant recruitment		19,769 (15,086-24,451)	0.48 (0.23-0.73)	27.24	0.56
	Random walk with trend	1.5 (1.08-2.1)		0.73 (0.35-0.98)	36.13	9.44
	Hockey-stick	2.4 (1.8-3.6)	19,769 (15,086-26,012)	0.48 (0.23-0.73)	30.41	3.73
	Ricker	6.9 (3.3-15)	22,890 (19,769-33,816)	0.48 (0.23-0.61)	26.69	0.00
	Beverton-Holt	25 (5.1-25)	21,329 (16,647-32,255)	0.48 (0.23-0.73)	30.99	4.30
Sandy River ^b	Random walk			0.73 (0.48-0.86)	17.21	0.00
	Constant recruitment		753 (506-1,247)	0.73 (0.35-0.98)	22.62	5.40
	Random walk with trend	0.94 (0.6-1.4)		0.73 (0.35-0.73)	21.37	4.15
	Hockey-stick	1.15 (0.6-2.7)	918 (588-1,494)	0.61 (0.35-0.73)	28.11	10.90
	Ricker	1.08 (0.65-12)	2,564 (671-2,564)	0.73 (0.23-0.73)	28.34	11.13
	Beverton-Holt	1.4 (0.75-25)	2,564 (835-2,564)	0.73 (0.35-0.86)	28.41	11.20

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference <2 are highlighted in light gray.

^b Recruits based on natural-origin escapement, not preharvest.

Table G.2.b Lower Columbia River spring chinook salmon population parameter estimates and model comparison. Except as noted, recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.) Exceptions to these spawner and recruit definitions occur because of data limitations.

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Cowlitz River ^b	Random walk			0.73 (0.61-0.86)	35.68	23.23
	Constant recruitment		315 (262-351)	0.35 (0.23-0.35)	12.45	0.00
	Random walk with trend	1.15 (0.85-1.5)		0.73 (0.48-0.86)	37.95	25.50
	Hockey-stick	3.3 (2.1-3.6)	315 (262-351)	0.35 (0.23-0.35)	15.62	3.17
	Ricker	2.1 (1.5-3.9)	422 (333-511)	0.48 (0.35-0.48)	26.60	14.15
	Beverton-Holt	25 (4.5-25)	333 (280-422)	0.35 (0.23-0.35)	15.79	3.34

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference <2 are highlighted in light gray.

^b Recruits based on natural-origin escapement, not preharvest.

Table G.2.c Lower Columbia River fall chinook salmon population parameter estimates and model comparison. Except as noted, recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners). Exceptions to these spawner and recruit definitions occur because of data limitations.

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Big White Salmon	Random walk			1.62 (1.24-1.87)	58.53	20.18
	Constant recruitment		901 (732-1,238)	0.73 (0.48-0.86)	38.35	0.00
	Random walk with trend	3 (1.8-5.1)		1.11 (0.86-1.24)	50.42	12.07
	Hockey-stick	6.9 (3.6-9.9)	1,069 (732-1,743)	0.73 (0.48-0.86)	40.39	2.04
	Ricker	6.9 (4.5-11)	1,575 (1,069-2,248)	0.73 (0.48-0.86)	41.00	2.66
	Beverton-Holt	19 (6.6-25)	1,238 (901-2,080)	0.73 (0.48-0.86)	40.54	2.19
Coweeman River	Random walk			2.12 (1.74-2.37)	62.37	31.30
	Constant recruitment		994 (734-1,254)	0.61 (0.48-0.73)	31.07	0.00
	Random walk with trend	6 (3.6-9.3)		1.11 (0.48-1.36)	46.21	15.14
	Hockey-stick	11 (7.8-25)	1,254 (864-1,774)	0.61 (0.35-0.61)	32.97	1.90
	Ricker	13 (9.6-20)	1,644 (1,254-2,294)	0.61 (0.35-0.61)	31.15	0.08
	Beverton-Holt	25 (13-25)	1,384 (994-2,684)	0.61 (0.35-0.73)	32.76	1.69
Cowlitz River	Random walk			1.24 (0.48-1.74)	44.25	12.16
	Constant recruitment		1,377 (872-2,051)	0.86 (0.48-0.98)	36.87	4.78
	Random walk with trend	0.6 (0.6-0.96)		1.11 (0.48-1.49)	44.74	12.65
	Hockey-stick	1 (0.75-1.8)	1,545 (1,040-2,388)	0.86 (0.35-0.98)	39.72	7.64
	Ricker	3.9 (1.5-8.4)	2,051 (1,545-3,566)	0.61 (0.23-0.73)	32.08	0.00
	Beverton-Holt	25 (1.8-25)	1,545 (1,040-3,735)	0.86 (0.35-0.98)	40.42	8.33

Table G.2.c cont.

East Fork Lewis River	Random walk			0.86 (0.61-1.11)	37.28	14.33
	Constant recruitment		573 (460-686)	0.48 (0.35-0.61)	22.95	0.00
	Random walk with trend	1.8 (1.4-2.4)		0.61 (0.35-0.73)	30.96	8.01
	Hockey-stick	3 (1.8-3.9)	573 (460-686)	0.48 (0.23-0.48)	25.10	2.15
	Ricker	5.1 (2.7-9.9)	630 (517-799)	0.48 (0.23-0.61)	26.03	3.09
	Beverton-Holt	15 (3.3-25)	630 (517-1,025)	0.48 (0.35-0.61)	25.98	3.03
Elochoman River	Random walk			1.36 (1.11-1.62)	51.04	12.38
	Constant recruitment		626 (432-819)	0.86 (0.61-0.98)	38.66	0.00
	Random walk with trend	1.8 (0.94-3)		1.24 (0.86-1.62)	51.44	12.78
	Hockey-stick	9.9 (1.8-10.5)	626 (432-1,109)	0.86 (0.61-0.86)	41.97	3.31
	Ricker	4.5 (2.7-6.9)	1013 (723-1,496)	0.73 (0.48-0.86)	39.51	0.85
	Beverton-Holt	25 (3.6-25)	626 (432-1,496)	0.86 (0.61-0.86)	41.88	3.22
Grays River	Random walk			1.49 (1.24-1.62)	55.62	0.00
	Constant recruitment		371 (141-716)	1.49 (1.24-1.74)	60.11	4.49
	Random walk with trend	1.3 (0.7-2.1)		1.36 (1.11-1.62)	57.87	2.26
	Hockey-stick	2.1 (1.08-9.6)	716 (371-1,176)	1.24 (0.98-1.36)	57.27	1.65
	Ricker	3 (1.2-7.2)	601 (371-1,865)	1.24 (0.86-1.36)	57.14	1.53
	Beverton-Holt	3.9 (1.3-12)	716 (371-3,359)	1.24 (0.98-1.36)	57.50	1.89
Kalama River	Random walk			1.36 (0.98-1.74)	51.46	12.57
	Constant recruitment		7369 (4,917-9,821)	0.86 (0.48-0.98)	38.88	0.00
	Random walk with trend	1.5 (0.85-3)		1.36 (0.86-1.62)	52.54	13.66

Willamette/Lower Columbia Salmonid Viability Criteria

Table G.2.c cont.

	Hockey-stick	6 (1.8-9.3)	7,369 (4,917-9,821)	0.86 (0.48-0.98)	42.19	3.31
	Ricker	3.9 (2.4-9.3)	9,821 (7,369-14,724)	0.86 (0.61-0.98)	44.49	5.60
	Beverton-Holt	25 (6.9-25)	7,369 (4,917-11,047)	0.86 (0.61-0.98)	42.59	3.70
	Random walk			1.11 (0.86-1.36)	45.87	4.91
	Constant recruitment		2,465 (1,389-3,542)	0.86 (0.48-1.11)	40.96	0.00
Mill Creek	Random walk with trend	1.5 (0.92-2.4)		1.11 (0.73-1.24)	46.60	5.65
	Hockey-stick	5.1 (1.2-11)	2,465 (1,748-4,618)	0.86 (0.48-0.98)	44.27	3.31
	Ricker	3.3 (1.5-7.2)	3,183 (2,465-5,695)	0.86 (0.48-1.11)	44.61	3.65
	Beverton-Holt	12 (1.8-25)	2,824 (2,107-10,001)	0.86 (0.61-0.98)	44.12	3.16
	Random walk			1.11 (0.73-1.36)	44.61	9.86
Washougal River	Constant recruitment		2,692 (2,000-3,383)	0.73 (0.35-0.86)	34.75	0.00
	Random walk with trend	1.2 (0.7-1.8)		1.11 (0.73-1.36)	47.07	12.32
	Hockey-stick	8.4 (1.4-12)	2,692 (2,000-3,729)	0.73 (0.35-0.86)	38.06	3.31
	Ricker	8.1 (3.6-16)	4,075 (3,037-5,458)	0.73 (0.35-0.86)	37.34	2.59
	Beverton-Holt	25 (25-25)	2,692 (2,000-4,075)	0.73 (0.35-0.86)	38.68	3.93
Wind River ^b	Random walk			1.99 (1.24-2.5)	44.95	14.79
	Constant recruitment		208 (112-351)	1.11 (0.61-1.49)	36.52	6.36
	Random walk with trend	1.4 (0.6-3.9)		1.99 (1.24-2.5)	47.87	17.72
	Hockey-stick	20 (2.1-20)	255 (112-446)	1.11 (0.48-1.49)	40.07	9.91
	Ricker	18 (9.9-25)	780 (494-1,018)	0.61 (0.23-0.86)	30.16	0.00

Table G.2.c cont.

Clackamas River ^c	Beverton-Holt	25 (25-25)	255 (112-494)	1.11 (0.48-1.49)	40.46	10.31
	Random walk			0.86 (0.73-1.11)	68.84	22.66
	Constant recruitment		447 (387-568)	0.61 (0.48-0.61)	46.18	0.00
	Random walk with trend	1.02 (0.75-1.4)		0.86 (0.73-1.11)	71.19	25.01
	Hockey-stick	2.7 (1.5-3.9)	508 (387-568)	0.61 (0.35-0.61)	48.09	1.91
	Ricker	3.3 (2.1-5.1)	568 (508-750)	0.61 (0.35-0.61)	47.76	1.57
	Beverton-Holt	20 (4.2-25)	508 (447-629)	0.61 (0.48-0.61)	48.85	2.66

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference <2 are highlighted in light gray.

^b Recruits based on natural-origin escapement, not preharvest.

^c Recruits based on natural-origin escapement, not preharvest. Spawners based on total spawners, and the fraction of hatchery-origin is unknown.

Table G.2.d Lower Columbia River winter steelhead population parameter estimates and model comparison. Except as noted, recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.) Exceptions to these spawner and recruit definitions occur because of data limitations.

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
East Fork Lewis River ^b	Random walk			0.98 (0.73-1.11)	14.83	3.19
	Constant recruitment		86 (82-88)	0.23 (0.23-0.23)	11.64	0.00
	Random walk with trend	0.6 (0.6-0.6)		0.48 (0.23-0.61)	21.15	9.51
	Hockey-stick	0.6 (0.6-1.08)	86 (82-90)	0.23 (0.23-0.23)	Infinity	Infinity
	Ricker	1.02 (0.65-1.4)	90 (84-95)	0.23 (0.23-0.23)	Infinity	Infinity
	Beverton-Holt	3.9 (1.02-25)	95 (84-132)	0.23 (0.23-0.23)	Infinity	Infinity
Clackamas River	Random walk			1.24 (1.11-1.36)	126.10	66.06
	Constant recruitment		4,152 (3,696-5,063)	0.48 (0.35-0.61)	60.04	0.00
	Random walk with trend	2.7 (2.4-3.3)		0.73 (0.61-0.73)	82.57	22.53
	Hockey-stick	7.5 (3.3-7.8)	4,152 (3,696-5,063)	0.48 (0.35-0.61)	62.41	2.36
	Ricker	6.6 (4.5-9.6)	5,063 (4,152-5,518)	0.48 (0.35-0.61)	66.14	6.10
	Beverton-Holt	25 (8.1-25)	5,063 (4,152-6,429)	0.48 (0.35-0.61)	62.93	2.89
Kalama River	Random walk			0.61 (0.48-0.86)	40.55	5.90
	Constant recruitment		1,108 (952-1,419)	0.48 (0.35-0.61)	34.65	0.00
	Random walk with trend	0.88 (0.7-1.15)		0.61 (0.48-0.73)	42.04	7.40
	Hockey-stick	2.4 (0.85-2.4)	1,108 (952-1,574)	0.48 (0.35-0.61)	37.50	2.85
	Ricker	2.1 (1.15-3.3)	1,263 (1,108-1,729)	0.48 (0.35-0.61)	36.70	2.05
	Beverton-Holt	5.1 (1.4-25)	1,419 (1,108-2,817)	0.48 (0.35-0.61)	36.79	2.15

Table G.2.d cont.

North Fork Toutle	Random walk		1.11 (0.35-1.74)	23.54	9.18
	Constant recruitment		173 (131-214)	0.35 (0.23-0.48)	14.36 0.00
	Random walk with trend	1.8 (1.1-3.3)	0.86 (0.23-1.24)	25.07	10.71
	Hockey-stick	9.6 (1.5-18)	173 (142-225)	0.35 (0.23-0.48)	21.36 7.00
	Ricker	4.5 (2.1-11)	235 (183-359)	0.61 (0.23-0.73)	26.47 12.11
	Beverton-Holt	25 (2.4-25)	183 (162-370)	0.35 (0.23-0.61)	21.63 7.27
South Fork Toutle	Random walk		0.48 (0.23-0.73)	10.71	0.00
	Constant recruitment		1,526 (1,224-1,828)	0.35 (0.23-0.35)	12.07 1.36
	Random walk with trend	0.94 (0.7-1.3)	0.48 (0.23-0.61)	17.31	6.59
	Hockey-stick	1.8 (0.7-20)	1,526 (1,299-1,903)	0.35 (0.23-0.35)	32.07 21.36
	Ricker	3.9 (0.83-9.9)	1,677 (1,526-3,186)	0.35 (0.23-0.35)	33.07 22.36
	Beverton-Holt	25 (1.15-25)	1,526 (1,375-3,186)	0.35 (0.23-0.35)	32.22 21.51
Sandy River	Random walk		0.23 (0.23-0.35)	4.92	8.98
	Constant recruitment		2,696 (2,616-2,855)	0.23 (0.23-0.23)	-4.07 0.00
	Random walk with trend	0.96 (0.83-1.08)	0.23 (0.23-0.35)	7.64	11.71
	Hockey-stick	1.3 (1.15-2.7)	2,775 (2,616-2,855)	0.23 (0.23-0.23)	0.01 4.08
	Ricker	2.7 (2.4-3.9)	2,775 (2,696-3,014)	0.23 (0.23-0.23)	0.07 4.14
	Beverton-Holt	25 (7.8-25)	2,855 (2,696-3,173)	0.23 (0.23-0.23)	0.27 4.33

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference <2 are highlighted in light gray.

^b Recruits based on natural-origin escapement, not preharvest.

Table G.2.e Lower Columbia River summer steelhead population parameter estimates and model comparison. Recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.)

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Kalama River	Random walk			1.49 (1.24-1.87)	76.34	26.72
	Constant recruitment		906 (687-1,343)	0.73 (0.61-0.86)	49.63	0.00
	Random walk with trend	0.6 (0.6-0.6)		1.24 (0.86-1.49)	68.72	19.09
	Hockey-stick	0.8 (0.6-2.7)	906 (687-1,343)	0.73 (0.48-0.86)	52.42	2.79
	Ricker	0.65 (0.6-1.3)	1,343 (906-1,781)	0.86 (0.61-0.98)	55.69	6.06
	Beverton-Holt	5.1 (0.83-25)	1,125 (906-1,562)	0.73 (0.48-0.86)	52.48	2.85
Washougal River	Random walk			0.61 (0.48-0.61)	20.84	7.45
	Constant recruitment		178 (151-218)	0.35 (0.23-0.48)	13.39	0.00
	Random walk with trend	0.8 (0.6-1.04)		0.48 (0.23-0.61)	21.87	8.48
	Hockey-stick	1.8 (0.7-8.7)	178 (151-272)	0.35 (0.23-0.48)	17.32	3.93
	Ricker	1.8 (0.92-3)	205 (178-299)	0.35 (0.23-0.48)	19.62	6.23
	Beverton-Holt	7.2 (1.1-25)	205 (164-489)	0.35 (0.23-0.35)	17.16	3.77
Wind River	Random walk			0.48 (0.35-0.61)	13.88	11.19
	Constant recruitment		486 (419-587)	0.35 (0.23-0.35)	11.50	8.81
	Random walk with trend	0.65 (0.6-0.7)		0.23 (0.23-0.23)	2.69	0.00
	Hockey-stick	0.65 (0.6-0.75)	855 (486-855)	0.23 (0.23-0.23)	7.82	5.12
	Ricker	0.75 (0.65-0.9)	1,290 (1,089-1,290)	0.23 (0.23-0.23)	8.20	5.50
	Beverton-Holt	1.02 (0.85-1.3)	1,290 (1,290-1,290)	0.23 (0.23-0.23)	9.63	6.93

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference < 2 are highlighted in light gray.

Table G.2.f Columbia River chum salmon population parameter estimates and model comparison.
 Recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.)

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Grays River	Random walk			1.11 (0.86-1.49)	84.72	14.88
	Constant recruitment		402 (319-569)	0.86 (0.61-0.98)	69.85	0.00
	Random walk with trend	1.3 (0.94-1.8)		1.11 (0.73-1.36)	85.37	15.52
	Hockey-stick	25 (1.15-25)	402 (319-1,069)	0.86 (0.61-0.98)	72.44	2.59
	Ricker	2.1 (1.4-4.2)	819 (569-1,152)	0.98 (0.73-1.24)	82.29	12.45
	Beverton-Holt	25 (1.8-25)	485 (402-1,152)	0.86 (0.61-0.98)	72.38	2.53
Hardy Creek	Random walk			1.49 (1.11-1.99)	145.19	51.33
	Constant recruitment		180 (149-212)	0.73 (0.61-0.98)	93.86	0.00
	Random walk with trend	1.4 (0.96-2.1)		1.49 (1.11-1.87)	145.21	51.35
	Hockey-stick	25 (7.8-25)	180 (149-244)	0.86 (0.61-0.98)	103.63	9.77
	Ricker	4.5 (2.4-7.8)	338 (244-433)	1.11 (0.86-1.49)	127.56	33.70
	Beverton-Holt	25 (9-25)	212 (149-244)	0.86 (0.61-0.98)	103.57	9.71
Lower Gorge	Random walk			2.12 (1.36-2.5)	227.65	100.53
	Constant recruitment		474 (383-565)	0.73 (0.61-0.98)	127.12	0.00
	Random walk with trend	1.5 (0.96-2.4)		2.12 (1.36-2.5)	227.97	100.85
	Hockey-stick	25 (25-25)	383 (383-474)	1.24 (0.73-1.49)	172.21	45.10
	Ricker	6.3 (3.3-16)	929 (656-1,658)	1.62 (0.98-1.99)	200.54	73.42
	Beverton-Holt	25 (25-25)	474 (383-565)	1.24 (0.73-1.49)	173.15	46.03

^a The a, b, and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference < 2 are highlighted in light gray.

Table G.2.g Upper Willamette River spring chinook salmon population parameter estimates and model comparison. Except as noted, recruits are based on estimates of preharvest natural-origin fish, and spawners are based on the estimate of natural-origin spawners plus half of the hatchery-origin spawners (hatchery-origin spawners are assumed to have lower reproductive success than natural-origin spawners.) Exceptions to these spawner and recruit definitions occur because of data limitations.

Population	Model ^a	a	b	σ^2	AICc	Relative AICc
Clackamas River ^c	Random walk			0.73 (0.61-0.86)	92.86	11.27
	Constant recruitment		1,238 (1,036-1,440)	0.86 (0.73-0.98)	102.00	20.41
	Random walk with trend	1.2 (0.99-1.5)		0.73 (0.61-0.86)	92.90	11.31
	Hockey-stick	1.5 (1.2-2.1)	2,250 (1,845-2,655)	0.61 (0.48-0.73)	82.03	0.43
	Ricker	2.1 (1.5-2.7)	2,048 (1,845-2,858)	0.61 (0.48-0.73)	81.59	0.00
	Beverton-Holt	2.1 (1.5-3.3)	3,465 (2,453-5,287)	0.61 (0.48-0.73)	82.67	1.08
McKenzie River ^c	Random walk			0.86 (0.61-0.98)	65.10	24.05
	Constant recruitment		2,242 (1,984-2,760)	0.48 (0.35-0.61)	41.05	0.00
	Random walk with trend	0.94 (0.75-1.3)		0.86 (0.61-0.98)	67.34	26.28
	Hockey-stick	3 (1.3-9.6)	2,242 (1,984-2,760)	0.48 (0.35-0.61)	43.62	2.57
	Ricker	2.7 (1.8-4.5)	2,760 (2,242-3,278)	0.61 (0.35-0.61)	50.62	9.56
	Beverton-Holt	25 (5.1-25)	2,242 (1,984-3,019)	0.48 (0.35-0.61)	44.62	3.57

^a The a , b , and σ^2 parameters for each model are described in Table G.1. The 95% confidence intervals on the parameter estimated are shown in parentheses. The AICc best approximating model for each population is highlighted in dark gray, and any models with an AICc difference <2 are highlighted in light gray.

^c Recruits based on natural-origin escapement, not preharvest. Spawners based on total spawners, and the fraction of hatchery-origin is unknown.

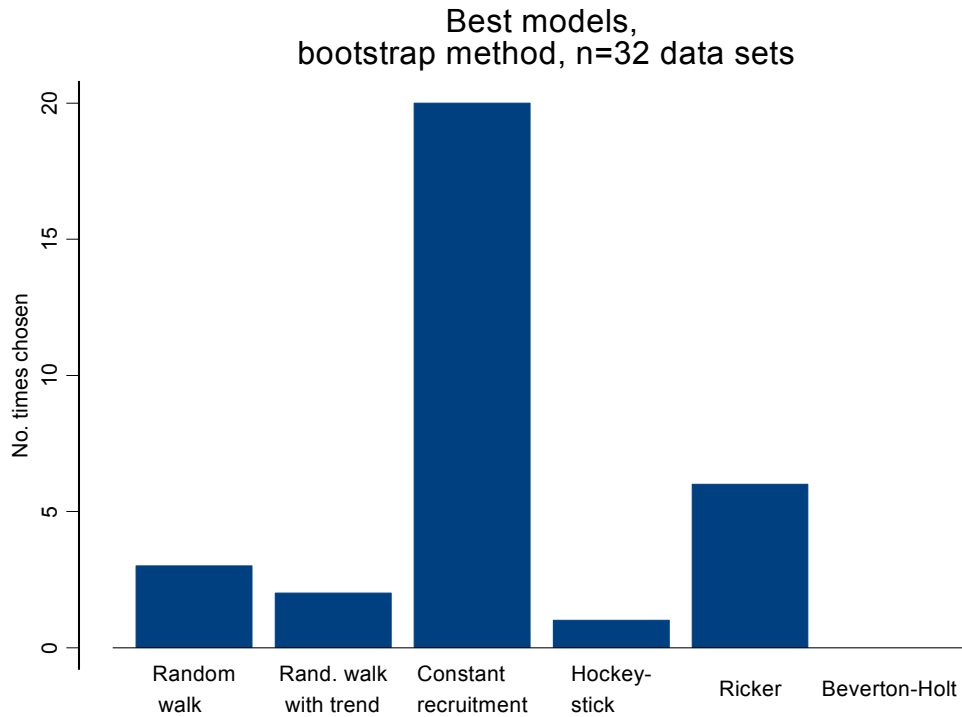


Figure G.9 Frequency of recruitment models selected as the best approximate models for 32 Willamette and Lower Columbia salmon populations. Models were selected using relative AICc method (see Table G.2).

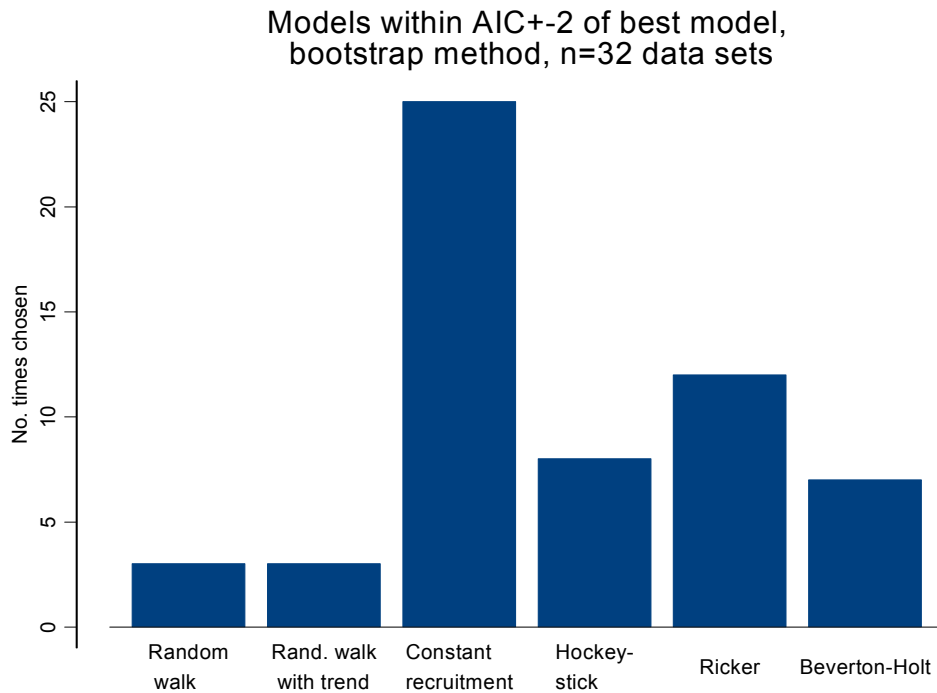


Figure G.10 Frequency of recruitment models selected as the best or near-best approximate models for 32 Willamette and Lower Columbia salmon populations. Models selected using relative AICc methods. Models considered near best had AIC difference values less than 2 (see Table G.2).

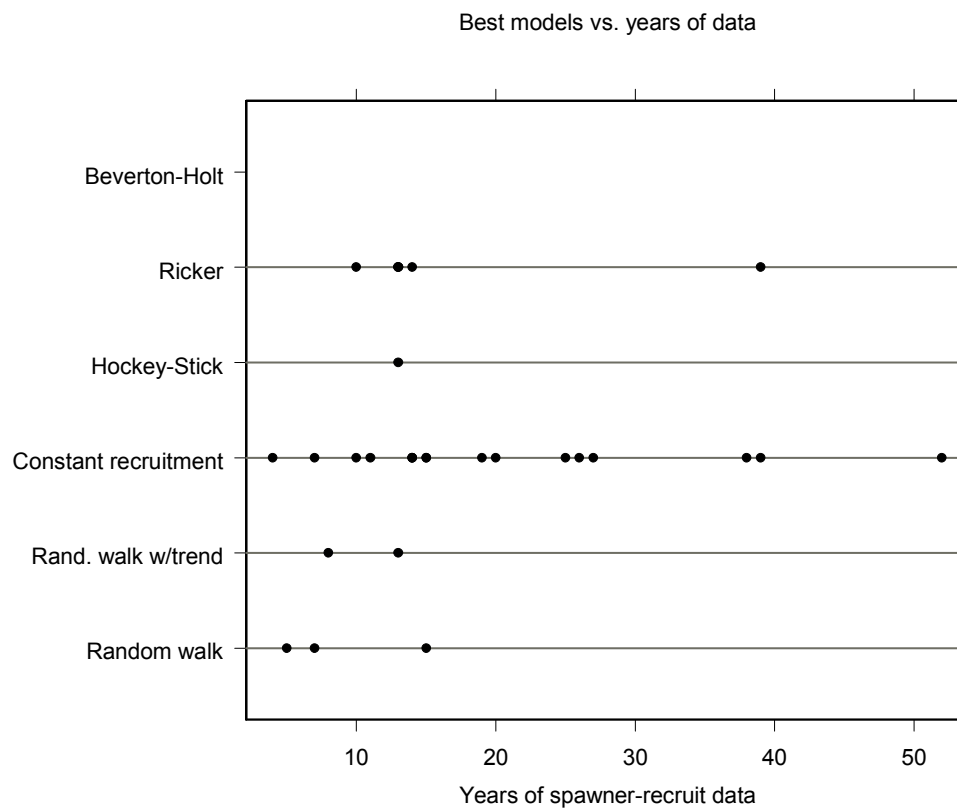


Figure G.11 Best model as a function of the number of years of spawner-recruit data (see Table G.2).

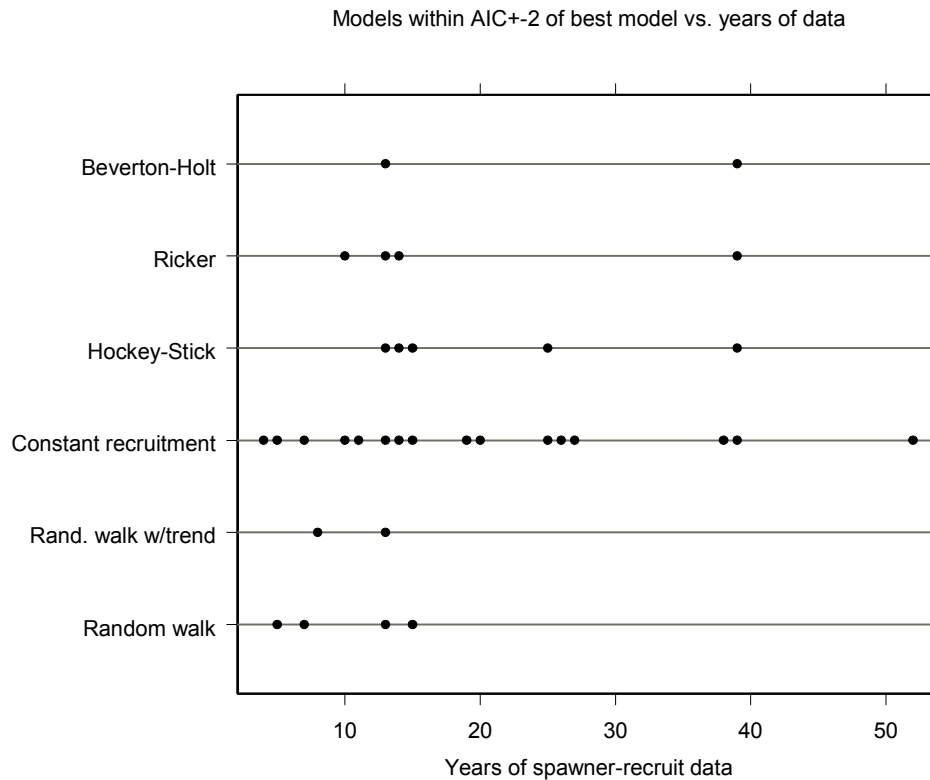


Figure G.12 Best model and near-best models as a function of the number of years of spawner-recruit data (see Table G.2). Models considered near best had AIC difference values less than 2.

Model Section Using Simulated Data

Although the observed data do not provide much information about the exact value of intrinsic productivity in a population, the selection of the constant recruitment model does suggest on the surface that productivity is greater than 1 and that the population is simply showing random fluctuations around a carrying capacity. However, this conclusion may be overly optimistic. We simulated a number of population trajectories using a hockey-stick model with an intrinsic productivity of 1. The populations were started substantially below the ceiling, so the trajectories were basically a random walk with an upper bound. We then calculated the recruits-per-spawner values from the trajectories, calculated parameters for the six models, and applied the AIC model selection approach (Figures G.11). In most of these examples, the best approximating model was either the constant recruitment model or the random-walk model. We speculate that the constant recruitment model is commonly selected as the best model because a short time series that samples a random-walk process appears as a cloud of points on a spawner-recruit graph. In the absence of data at very low (or very high) spawner numbers, the data are likely to fit a constant recruitment model. This is particularly true if there is any sort of population ceiling that leads to a flattening of the spawner-recruit cloud.

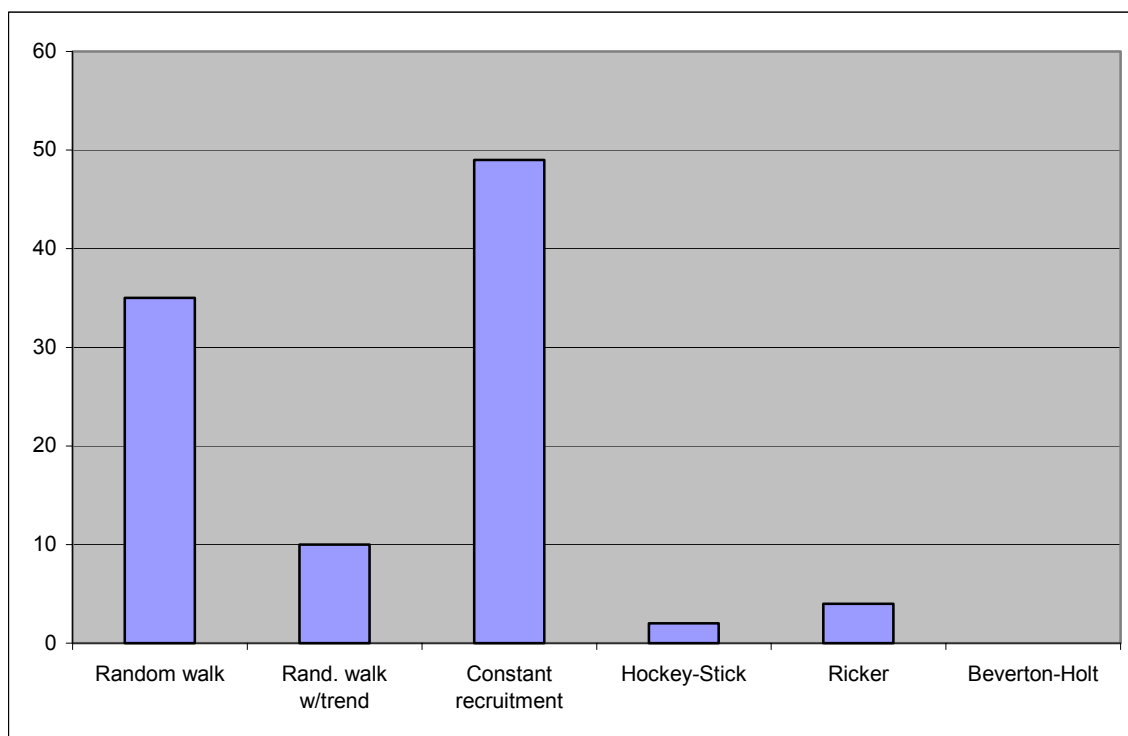


Figure G.11 Frequency of models selected as the best approximating model for simulated population trajectories. We simulated 100 25-year population trajectories with a hockey-stick model with an initial population size of 5,000, carrying capacity of 50,000, intrinsic productivity of 1, with log-normal process error (normal distribution mean = 0 and variance = 0.6). The variance of 0.6 is similar to that observed for Willamette/Lower Columbia populations (see Table G.2). The viability curves were generated for a semelparous population where the average percentages of individuals spawning at a given age are: age 1 = 0%, age 2 = 1%, age 3 = 19%, age 4 = 57%, and age 5 = 23%. This life-history structure is typical to that observed for chinook salmon.

Conclusions

Analysis of both the WLC populations and simulated trajectories suggest that the adult recruits-per-spawner data typically available for salmon populations will be inadequate to estimate intrinsic productivity. The lack of data at small spawner abundances make recruit curve-based productivity estimates highly uncertain.

Recruitment models are a foundation of harvest modeling and have been proposed as metrics for viability criteria. Before applying these models, it is important to have a solid understanding of the uncertainties involved in parameter estimation and model selection.

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APPENDIX H

AN APPROACH TO POPULATION VIABILITY ANALYSIS FOR STEELHEAD AND SALMON USING STOCK-RECRUITMENT CURVES WITH MULTIPLE LIFE STAGES

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Overview

The results from Appendix G suggest that fitting recruitment curves to adult recruits per spawner data is unlikely to provide precise estimates of intrinsic productivity for salmon populations that could be used for viability criteria. This may be due to measurement error, environmental variation within the data, and/or the addition of hatchery spawners to naturally spawning populations so that few populations have low enough spawning escapements to accurately measure intrinsic productivity. However, precision in estimating intrinsic productivity may potentially be increased by partitioning the life cycle into freshwater and marine phases. This appendix explores an approach to setting viability criteria in which density dependence is assumed to occur in the freshwater life stage, and marine survival is considered a density-independent factor driven by the environment. Criteria with this approach would likely be a demonstration that, based on the assumption that habitat in the freshwater environment is stable, a population has sufficient freshwater productivity and capacity to persist in the face of hypothesized future marine survival patterns. With this approach, there is no single freshwater productivity, and capacity target, as multiple combinations of productivity and capacity, could produce identical extinction risks. Instead of a single *a priori* target, the approach could potentially be used retrospectively to evaluate whether a population has improved such that it has an acceptably high probability of persistence.

The approach in this appendix is applied to evaluate the current status of the Wind River steelhead population. Model parameters were fitted using the Wind River data, and a forward project of abundance was modeled under a number of hypothesized future ocean survival patterns. The Wind River data have not yet been analyzed using a formal model selection procedure like that described in Appendix G. There is concern about the level of applicability of this for this approach because few populations in the Willamette-Lower Columbia (WLC) domain have intensive smolt monitoring programs. Also, the Wind River data set is very short, and some WLC Technical Recovery Team (TRT) members are concerned that apparently better fit of the multiple life-stage models may be an artifact of simply adding another parameter (additional parameters almost always lead to better fit, but may not actually contribute to the utility of the model). However, the approach is promising. Assuming the appropriate data are collected and robust predictions can be made about future patterns of marine survival and

freshwater habitat, the approach provides a potential alternative to the population change criteria approach.

The Model

This approach uses a spawner-recruit relationship (SRR) to assess extinction risk (Chilcote 1998, Chilcote 1999, Routledge and Irvine 1999, and Johnson et al. 2000). Salmonids often demonstrate Ricker, Beverton-Holt, and/or hockey-stick SRR relationships. These SRR relationships assume density-independent mortality; that is, at low spawning densities, survival is independent of stock size (Hilborn and Walters 1992). However, it is unlikely that a population can continue a high rate of reproduction as the stock continues to increase. Eventually, available resources limit populations, and some reduction of recruits per spawner is observed as the spawning stock size increases. The Beverton-Holt and hockey-stick models assume that the quality and quantity of freshwater spawning and rearing habitat limits freshwater production of juvenile salmonids (Beverton and Holt 1957, Barrowman 2000). Unlike the Beverton-Holt and hockey-stick models, the Ricker model assumes declining recruitment at higher stock sizes. For the freshwater relationship the SRR can take any one of these forms, but for simplicity this overview will work with the Beverton-Holt form. The Beverton-Holt model is fit to the spawner and smolt data using maximum likelihood methods and assuming lognormal error (Hilborn and Walters 1992). As smolts enter the marine environment I assumed they had density-dependent survival (Ward 2000, Johnson et al. 2000, Emlen et al. 1990). Pearcy (1992) summarized data for anadromous fish that suggested the variation in marine survival often exceeds the variation in freshwater survival. Cramer (1996) confirmed that considerable variation is in the marine environment by examining hatchery coho and chinook salmon returns. This combination of density dependence in freshwater and density independence in the marine environment can be expressed in a stock-recruitment equation using a Beverton-Holt model as:

$$R = (\alpha S / (1 + \alpha S/\beta)) * m \quad (1)$$

where

R= the number of adults,
 S = the number of spawners,
 α = the freshwater intrinsic productivity of the stock,
 β = the freshwater carrying capacity of the stock, and
 m= marine survival.

Since steelhead are iteroparus, another term was added to account for repeat spawners. In SRR analysis, it is often assumed that environmental factors are constant over time. However, conditions such as floods during incubation and summer low flow have all been shown to alter the survival of a cohort. To account for this environmental variation, another term was added to the SRR, where e^{ϵ_t} was drawn from a normal distribution, $N(0, \sigma)$. The final added term is to ensure that the random error has a mean of 1.

$$R = (\alpha S / (1 + \alpha S/\beta)) * m * c(S_{t-1}) * e^{\epsilon_t} * e^{-(\sigma^2/2)} \quad (2)$$

where

- S_{t-1} = spawners from the previous year,
- ϕ = the repeat spawner rate,
- e_t = a normally distributed variable with a mean of zero,
- $(\sigma^2/2)$ = process error of the model fit.

These equations are incorporated into an age structure model using freshwater and ocean-age steelhead data from the Wind River. This approach incorporates realistic levels of parameter uncertainty in the freshwater and marine environments. If the smolt production is not limited under favorable ocean conditions, smolt production can be unrealistic. Therefore, an upper bound on smolt production was needed. For the Beverton-Holt model, I chose to limit production to 1.5 times the capacity. This has a limited effect on extinction risk since extinction would most likely occur under poor ocean conditions in which the predicted smolt production would not be capped.

I initiated the age-structure model at 500 spawners, the recent mean escapement. The simulations ran for 100-plus years. To estimate extinction risk I recorded the lowest population size, then recorded the number of times the population dropped below a quasi-extinction threshold. Chilcote (1999) suggested a quasi-extinction threshold of 150 spawners for small steelhead populations and 300 for larger steelhead populations. McElhany (2002) suggests quasi-extinction thresholds of 50. Based on the results of 100 simulations the extinction risk is estimated to be the percentage of time a population dropped below each quasi-extinction threshold. Alternative expressions can be adapted for the hockey-stick and Ricker models.

The underlying approach for pursuing life-cycle SSR modeling is that some of the variation in population abundance can be explained by fluctuations in run size and changes in marine survival. Figures H.1 and H.2 demonstrate why this may be true for the WLC domain. The data for Trout Creek steelhead demonstrate that smolt abundance is related to the abundance of female spawners (Figure H.1). The Kalama River data show strong correlation between steelhead run size and smolt-to-adult survival (Figure H.2). Since the goal of this approach is to estimate population persistence, future estimates of marine survival must be incorporated into the model. Five approaches were considered for capturing the future variation in marine survival. The first is that marine survival is random and does not follow a pattern. However, salmon populations have been documented to have periods of higher and lower productivity (Mantua et al. 1997). Anderson (1998) documented these periods for the catch of Columbia River chinook salmon. Due to the change in productivity over decadal scales, it is unlikely that random variation is likely occurring, and this option was not pursued.

The second approach was that marine survival followed a step function, with periods of poor ocean productivity followed by periods of favorable ocean productivity. Hare (1999) indicated this phase shift worked well for explaining the variation in Bristol Bay sockeye salmon SRR. In the productive period the intrinsic productivity on these sockeye stocks was twice that measured during the unproductive periods. However, in a poor or favorable marine survival pattern there is variation. To account for this, I set the variation to randomly range from the lowest to half of the high plus low during the poor period to the highest to half high plus low during the favorable period. Future ranges of marine survival were generated by randomly generating survivals within this range.

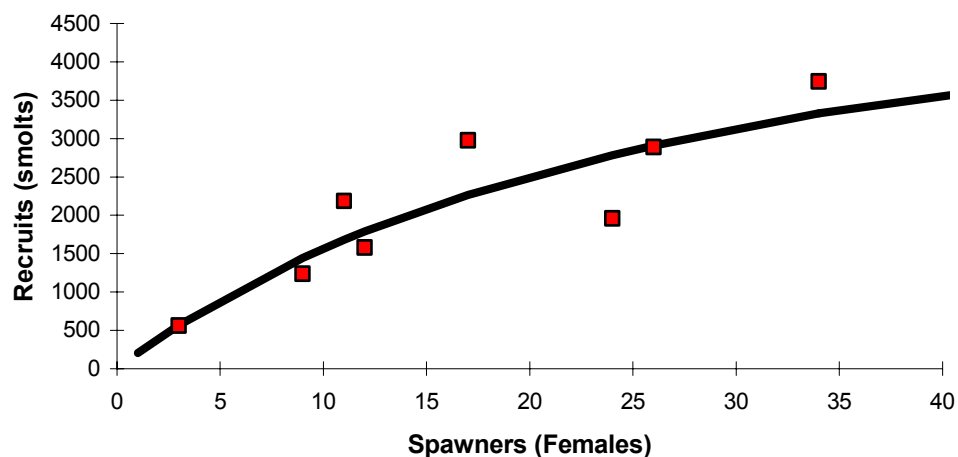


Figure H.1 Beverton-Holt model fit to wild steelhead data on Trout Creek, a tributary to the Wind River. This demonstrates that much of the variation in smolt abundance is explained by the number of spawners.

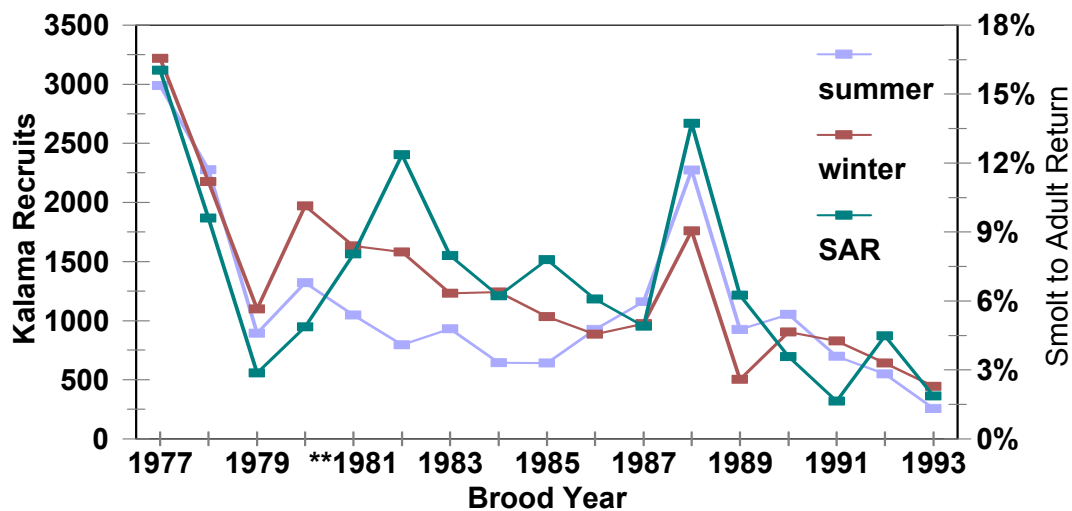


Figure H.2 Correlation Kalama River wild steelhead returns and hatchery smolt-to-adult survival rates in the Kalama River. The increase in the 1981 and 1982 broodyears SAR (smolt-to-adult return) was due to increased straying of Cowlitz and Toutle hatchery steelhead into the Kalama River.

The third approach was to use the Pacific Decadal Oscillation (PDO) as a surrogate for marine survival. The lowest observed smolt-to-adult survival was set to the annual PDO estimate that was the lowest, and the highest marine survival was set to the highest annual PDO. The simulation assumed the PDO repeated itself every 100 years. While strong correlations have been exhibited with the PDO for some salmonids, flows and hydroelectric operations in the Columbia River and local estuarine conditions may have a strong influence on survival as well.

A fourth approach was to use a sine wave to simulate cycles in ocean productivity. Hare et al. (1999) suggested that ocean cycles are decadal, and the PDO had a period of approximately 25 years. I calculated the highest and lowest observed smolt-to-adult survival, added the high and low, and divided by 2 to estimate the mean. The period on the sine wave was the mean minus the low.

Salmon populations face a higher level of extinction risk during periods of low ocean productivity. The most conservative approach was to set the future smolt-to-adult survival to the lowest observed in the data set.

Results

A Beverton-Holt model was fit to the Wind River steelhead SRR data (Figure H.3) Due to small sample size ($n = 8$) for Wind River steelhead, the original data point in the series was not included because there was uncertainty in the smolt-trapping data (the trap was out from April 22–May 5 due to damages from a flood), and the intrinsic productivity fit with all eight points led to a potentially unrealistic range observed for this species. However, I used all eight points to estimate σ . The most conservative estimate of smolt-to-adult survival for Wind River steelhead was 0.8%. In all simulations this led to a population of 0 fish in about 100 years (Figure H.4). Under this scenario, the probability of population persistence was 0%. When marine survivals were fixed at less than 1.2%, the population was likely to be less than the quasi-extinction threshold of 50 spawners (Figure H.5). The sine wave results indicate that when the smolt-to-adult survival ranges from 0.8% to 6% the probability of dropping below 50 adults is low. More simulations are needed to fully develop this criterion.

Discussion

A critical assumption of this or any population viability analysis (PVA) is stationarity; that is, that the environment in which the past data was collected is the same or very similar to the future condition being modelled. If this is not the case, there is no scientific basis for predicting future population sizes, unless one can accurately describe the future environment used by salmon and the new relationship between salmon and their future environment. Salmon abundance and productivity are correlated to freshwater, estuarine, and ocean habitat.

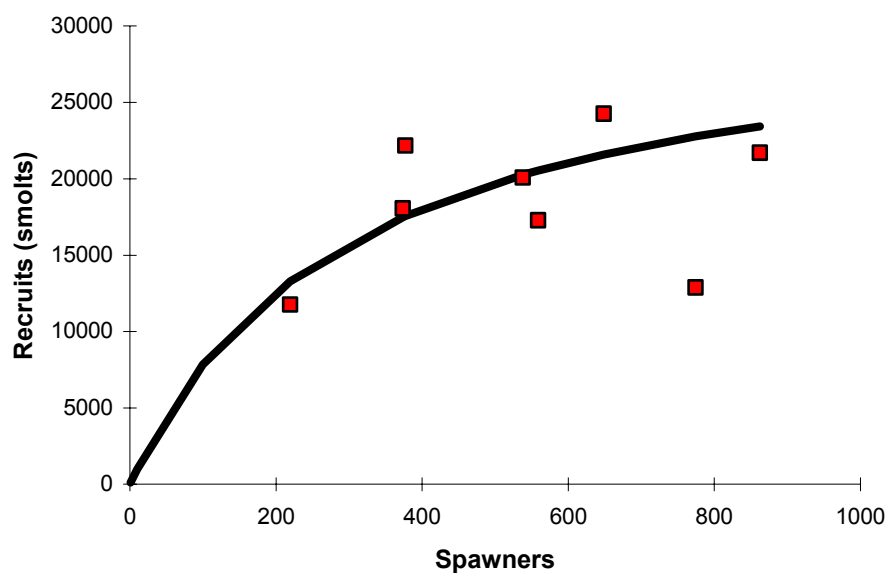


Figure H.3 A Beverton-Holt fit for Wind River steelhead, broodyears 1994–2000.

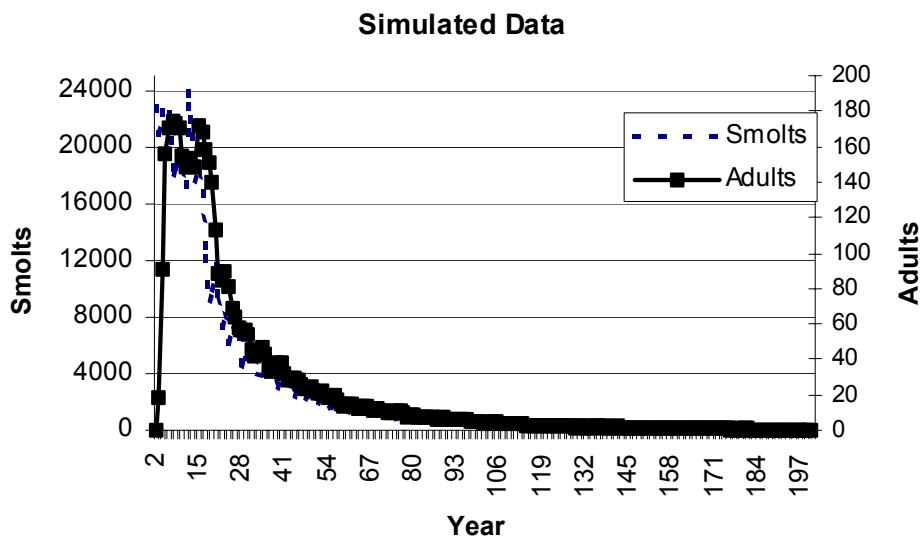


Figure H.4 Abundance of Wind River steelhead adults and smolts under lowest marine survival (0.8%). Modeling projects 0 adults in 100 years.

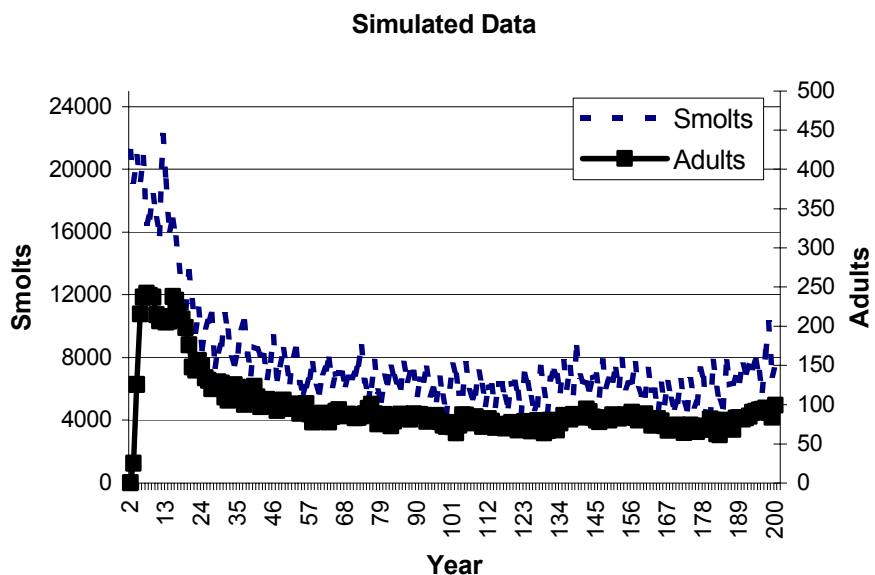


Figure H.5 Abundance of Wind River steelhead adults and smolts under a marine survival of 1.2%. The lowest estimated population size was 64 adults.

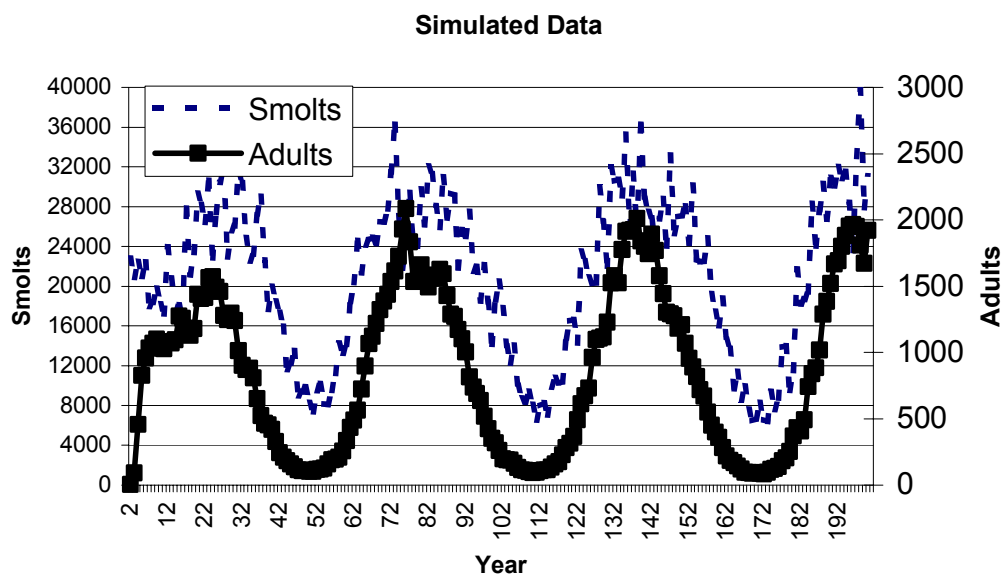


Figure H.6 Abundance of Wind River steelhead adults and smolts under 60-year ocean cycle with lowest marine survival (0.8%) and highest. Model predicts the minimum abundance will be 69 spawners.

Understanding how habitat is likely to change and the fish's response to this change would improve the predictability of any PVA. For this PVA to be successful, salmon habitat over the measuring period and over the forecast period must be consistent, there must be "no net loss in habitat." Furthermore, the range in ocean productivity over the measurement period must accurately reflect the future (i.e., no global warming). Therefore, to ensure that any PVA modeling is forecasting correctly, there must be an adequate assessment of existing habitat regulations to ensure stationarity into the future. Without this constant freshwater habitat the predictive ability of all PVA modeling is called into question.

The quasi-extinction threshold is very important because all PVA are very sensitive to this parameter. The concept of a quasi-extinction threshold can be simply stated: When a population drops below this level the chance for extinction is very high due to loss of genetic diversity, the ability to find a mate, and increased catastrophic risk. McElhany (2002) identified 50 spawners as the quasi-extinction threshold. However, a population that persists at 50 spawners for long periods of time will be at increased extinction risk due an erosion of genetic diversity.

Another parameter that PVA models are sensitive to is the intrinsic productivity or productivity at low spawning densities. If, at low densities, populations can produce many recruits, they show resiliency. Populations with high resiliency have a higher probability of persistence than populations with lower resiliency. However, Appendix E indicates that this parameter may be difficult to estimate unless a life-cycle approach is used. The data required for life-cycle modeling are not usually collected in current salmon or steelhead population monitoring programs.

The utility of PVA models is increased if they provide a constant target, determine population status prior to listing, provide information that assists in identifying factors for decline, and assess individual or a suite of actions in a recovery plan. This approach tries to address these concerns. When the various biological review teams (BRTs) recommended to list chinook, chum, and steelhead within the Lower Columbia River domain, there was considerable disagreement about whether populations should be listed, and whether they should be listed as threatened or endangered under the Endangered Species Act (ESA). There was no quantitative analysis of individual populations regarding their viability, which was appropriate since the BRT was concerned with assessing evolutionarily significant unit (ESU) risk. Given the range of professional opinions among the BRT members, it is likely that some populations within the ESU were healthy at the time of listing. Therefore, the PVA model should not make *a priori* assumptions about population status—that all populations need to grow to be viable. The model should independently assess the population's status without knowing how other models assessed its status. This approach is important because it then becomes a tool for assessing the status of a population for a listing decision.

This type of PVA was in direct response to comments received from the first draft. The Wind River data were included as an example of an alternative approach to the population change criteria (McElhany 2002). The Wind River data set is not of sufficient duration to fully estimate extinction risk. In addition, data points higher and lower than the eight observed would better define the freshwater relationship over a broader range of conditions. Data sets that encompass 20 years if they cover the range in environmental conditions should be considered in this or similar PVA approaches.

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APPENDIX I

BROAD-SCALE HABITAT ANALYSES TO ESTIMATE FISH DENSITIES FOR VIABILITY CRITERIA

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Background

In this appendix, we describe a method for estimating quantities of currently and historically available habitat and for estimating fish densities implied by a range of viability criteria. We summarize results for the Willamette Lower Columbia (WLC) recovery domain. The rationale for examining implied fish densities for both currently and historically available stream miles is to consider the implications of a range of viability criteria on specific populations. These analyses can be used to answer the following questions:

- How much potential spawning or rearing habitat is currently accessible in particular watersheds for a given species of interest?
- How much potential spawning or rearing habitat might once have been available in particular watersheds for a given species of interest?
- Is a viability criterion reasonable given currently accessible habitat?
- Is a viability criterion reasonable given all historically available habitat?

The first step for many habitat-related recovery analyses is to estimate the amount of currently accessible and historically accessible habitat. These analyses provide an initial estimate of those quantities. One of the largest and most easily quantified anthropogenic changes to habitat quality or quantity has been the construction of large numbers of barriers to fish passage. Streams currently blocked to anadromous passage by a man-made barrier were historically available for spawning and rearing by multiple salmonid species. The first step in this analysis is to quantify the amounts and types of habitat that have been lost due to man-made barriers as a metric of habitat change. The results of this step form the building blocks for multiple additional analyses and are reported in detail here. The next step is to identify those habitat areas, currently and historically available, that are likely used by a particular species at a particular life stage. This classification might be based on mainstem versus tributary habitat, channel gradient, or other landform or land-use variables. The final step of our analysis is to calculate fish densities that would be necessary to meet potential, population-specific viability targets.

Methods

Stream Generation

One of the primary objectives of this analysis is to assess relative quantities and distribution of multiple types of habitat throughout the WLC domain. Because of the broad geographic range of the analysis, we required a Geographic Information System (GIS) stream network that was consistently available across all watersheds, but still of a fine enough resolution to discern differences in lengths of mainstem and tributary streams of varying widths. We determined that a 1:100,000 stream network (StreamNet 2001a) was not adequate to capture the stream features and measurements of interest. However, the available 1:24,000 stream networks were incomplete and used inconsistent methodologies.

We chose to generate 1:24,000 stream networks from 10-m drainage-enforced digital elevation models (DEMs), digital representations of three-dimensional terrain. Delineation and extraction of stream channel networks from DEMs is a well-represented procedure in practice and model development (Jenson and Domingue 1988, Tarboton et al. 1991, Montgomery and Foufoula-Georgiou 1993, Tarboton 1997). This technique uses drainage direction and flow accumulation (using slope and aspect from the DEM) across the landscape to delineate primary drainage and stream routes. Intuitively, the procedure can be described as estimating the location of the stream network by simulating the flow of water across the landscape. We chose a suite of programs called NetStream (Miller 2003) to assist us in generating a GIS-compatible, 1:24,000 spatial stream network. NetStream optimizes the resolution of low-order channels and can break the stream into topographically homogeneous segments or reaches based on DEM-derived valley width and channel gradient.

We generated streams for 17 fourth-field hydrologic units within the WLC domain. The watersheds were chosen based on distribution of Endangered Species Act (ESA) listed salmonid species and the extent of distinct populations (Myers et al. draft) within each evolutionarily significant unit (ESU). We performed quality control on the stream networks using other GIS stream network data sets. Stream channels were adjusted or regenerated where necessary. The NetStream program modeled approximately 111,780 km of streams at this scale for the 17 watersheds of interest. These were broken into 1.8 million stream segments or stream reaches. Reach length was 50 to 300 m, with a mean of 76 m. Reach length varied with upstream drainage area; larger rivers have larger drainage areas and less geomorphic variation so typically have longer reach lengths (Miller 2003). For example, in the Mollala-Pudding watershed, the model generated 4,108 km of stream, broken into 59,801 stream-reach segments. The same watershed at a lower resolution (1:100,000) is represented by only 1,708 km of stream. Because segments were identified based on tributary junctions and homogeneity in channel gradient, they are very similar in size and character to stream reaches that might be identified during field surveys. The similarity to field-derived reaches, and the small maximum size of the stream segments, is a strength of this analytical approach; habitat characteristics estimated for each reach will not be averaged over a long heterogeneous length of stream.

Computing limitations made it necessary for us to reduce both the number of stream segments and total segment length in the analysis. We clipped the drainage network to include only contiguous stream segments with a channel gradient of less than 20%. This clipping, or “pruning,” of the drainage network reduced the number of stream segments by about 50% while

effectively removing only the smallest-order streams, which are rarely, if ever, used by anadromous fish (Washington Forest Practices Board 2000). All further analyses were conducted on the pruned network.

NetStream calculated additional stream and habitat attributes based on the 10-m DEM (Table I.1). Channel gradient and valley floor widths were estimated for each segment. Given channel width (see next section), valley floor width can be used to determine whether the channel is constrained or unconstrained. Valley side-slope was calculated separately for each side of the river, averaging the gradient over the nearest 10 m from the edge of the channel and the nearest 100 m from the edge of the channel. (For further details on the NetStream program, see Miller 2003.) In the future, valley side-slope gradient and additional debris-flow modeling functionality in the NetStream program may be used to predict channel-bank stabilization measures and probability of landslides or debris flows. Modeled attributes not used in our immediate analysis may be used in future efforts to link landscape processes to in-stream habitat features and in-stream habitat conditions to fish densities.

Width Modeling

Fish use of habitat types is expected to vary by stream width. For example, pools in mainstem habitats might be expected to have higher densities of juvenile chinook than pools in tributaries. Several research groups have had good success modeling stream width using a combination of basin areas, channel gradients, and annual precipitation (Miller et al. 1996, Holsinger 2001, Hyatt et al. 2002, Clarke 2001).

Using simple linear regression analysis, we built a series of watershed-specific, stream-width models using existing habitat survey information from the Oregon Department of Fish and Wildlife (ODFW) Aquatic Inventories Project (AIP) (ODFW 2001a, Moore et al. 1999). These data sets contained measured channel width, as estimated at bankfull stage, for a total of 3,142 stream reaches. Reaches sampled by ODFW were 6,781 m long on average. Surveyed reaches were distributed fairly well throughout the Willamette and Lower Columbia watersheds in Oregon, and were available for 13 of the 17 fourth-field watersheds of interest. There was a greater emphasis on the smaller tributaries in the Willamette Basin (Figure I.1). We were unable to include data for watersheds on the Washington side of the Columbia River or on U.S. Forest Service lands because of data compatibility issues. Potential predictor variables for the stream-width models were from remotely sensed data and included basin area, average precipitation, and channel gradient. We plan to improve our stream-width modeling with recently updated AIP stream-reach data available for six of these watersheds from ODFW.

Stream width was modeled separately for the 13 watersheds with available data and for the entire WLC domain at once. Model fit varied dramatically between basins as a result of the distribution of field-surveyed reaches within the basin ($0.15 < R^2 < 0.76$). Model fit for all watersheds combined was adequate ($R^2 = 0.41$) but not as strong as identified in other smaller-scale or more field-intensive efforts. In most watersheds, stream reaches with a basin area smaller than 1 km² were removed from the analysis because they had such a strong effect on estimated parameters that the model did not fit the larger streams well. All watershed models included basin area, most included precipitation, and a few included channel gradient (Table I.2). Several models also included a multiplicative interaction between basin area and precipitation. A significant interaction term suggests that the effect of precipitation on stream width varied for

smaller versus larger streams. In watersheds where the watershed-specific model fit was worse than the overall model fit, we used either of two alternative models. If there was an adjacent watershed within the same ecoregion with a better model fit and similar geology and topography, we used the model for that adjacent watershed. Where there were no adjacent watersheds with similar attributes and better model fits, we used the overall model to predict stream width in the watershed with a weak model fit.

We are in the process of testing the stream-width models. We plan to use a Monte Carlo approach to estimate predictive uncertainty. We may also be able to use the distribution of survey reaches to correct model predictions. In the interim, we are dividing the modeled widths into four categories based on similar work conducted in Puget Sound (Beamer 2001). These categories are large main stem (> 25 m), small main stem (10–25 m), large tributary (5–10 m), and small tributary (< 5 m). Initial results are presented in Figure I.2.

Barriers

As in much of the Pacific Northwest, one of the major habitat alterations in the WLC domain has been the placement of barriers to upstream and/or downstream migration. Barriers—including dams, diversions, and culverts—can partially or fully block fish passage. We collected information on in-stream barriers to fish migration in order to identify accessible and inaccessible river segments. We identified and coordinated nine digital databases containing information on the location and passage of natural and anthropogenic barriers (summarized in Table I.3). Because the data were compiled from multiple sources, the positional accuracy and passage information varied. Originally, there were over 10,000 potential barriers in the combined data set. After removing duplicates, clipping data to our watershed boundaries, and removing barriers located on streams not present in our stream network, we included over 2,600 potential barriers. To estimate habitat changes resulting from barriers, we identified all river segments in our analysis as accessible, inaccessible due to a man-made barrier, or inaccessible due to natural barriers (Figure I.3).

We used a variety of methods to categorize barriers as passable, impassable, or partially passable. These included GIS coverages of current fish distribution (Streamnet 2001b), maps of historical fish distribution, nonspatial databases, personal communications with state agencies, and (since height was one attribute available for most of the barriers) published limits to fish passage by barrier height (Myers et al. 2002; Aaserude 1984). Uncertainty in these classifications remains because positional inaccuracy of some barriers prevented us from associating them with the stream network (even though they may be barriers to fish passage), and because many barriers do not have complete passage information (Table I.3). Where passage was unknown or incomplete, we classified the barrier as passable; therefore, our analysis represents a conservative estimate of inaccessible stream habitat for most watersheds. Once the barriers were classified, all downstream and upstream stream segments were identified with the appropriate barrier passage codes, which were then summarized to simplified stream accessibility codes, as represented in Figure I.3. Individual barriers within the WLC block from 1 to 2,000 km of stream each. Local biologists reviewed the stream accessibility maps; they were requested to indicate erroneously classified streams within their respective geographic regions of expertise. Reviewers were asked to focus primarily on map errors (> 3 km) likely to impact general, broad-scale accessibility

ratios for the watershed. All changes indicated by the biologists were incorporated into the final maps and summaries (Nusum 2002, Meyer 2002, Shively 2002, Stearns 2002¹).

Future work conducting field inventories and developing statistical techniques for this type of spatial data will be required to reduce and display the uncertainty in our analysis. We also plan to incorporate newly available data. Spatial data (at a scale of 1:24,000) on fish distribution and passage and locations of culverts and dams has recently become available for Washington State (SSHIAP 2002) and has been incorporated in the final stream accessibility data. Updated fish distribution and barrier passage information has recently become available in Oregon (ODFW 2001, BLM 2000). These data will also be incorporated for specific watersheds as required by future studies. Eventually, we will quantify and display any remaining uncertainty about whether each stream segment is accessible to each species of concern.

Identification of Prime and Possible Habitat Attributes

Fish use of particular stream reaches is based on a host of habitat characteristics including nutrient status, channel gradient, substrate, cover, flow, water depth, and channel width. For this large-scale analysis, we needed to identify those areas most likely to be used by fish based on habitat characteristics we could identify from data available over the entire WLC. We surveyed eight local fisheries biologists to identify the suite of habitat characteristics that might indicate prime or possible habitat for each listed species at each relevant life stage. For steelhead and chinook, channel gradient was the best available habitat characteristic for classifying stream reaches in a way that would suggest whether the reach might be used for spawning and/or rearing. Ideally, we would have been able to use several of the other measured or modeled characteristics such as channel width or riparian vegetation; however, no quantitative thresholds could be determined for these characteristics. By combining local biologists' responses, we created a series of gradient thresholds describing prime and possible spawning and rearing habitat for each species (Figure I.4, Table I.4). For chum salmon, we identified a gradient cutoff to use initially (Table I.4) and a set of rules based on channel width and distance upstream from a tributary junction that can eventually be used as the basis of a better classification system. Further spatial analyses and programming will be required to implement these chum classification rules using our modeled channel width. While this is an extremely rough guide, it does help us to eliminate from our analyses those segments that are much less likely to be used by a particular species at a particular life stage.

¹ Nusum, M. 2002, personal communication—map review, Oregon Department of Fish and Wildlife South Willamette Watershed District Office, 7118 NE Vandenberg Avenue, Corvallis, OR 97330. Meyer, K., 2002, personal communication—map review, Cowlitz Valley Ranger District Fisheries Biologist, Gifford Pinchot National Forest, 10024 US Hwy 12, Randle, WA 98377. Shively, D., 2002, personal communication—map review, Fisheries Program Manager, Mt. Hood National Forest, 16400 Champion Way, Sandy, OR 97055. Stearns, C., 2002, personal communication—map review, Fisheries Biologist, Washington Department of Fish and Wildlife, 600 Capitol Way North, Olympia, WA 98501-1091.

Results

Available Stream Kilometers by Population

Currently and historically accessible, prime and possible stream kilometers are summarized for each population in the WLC ESUs in Tables I.5 through I.19. Summaries in these tables represent those reaches that meet prime or possible spawning criteria and include streams to the upper limits of the clipped stream coverage. These streams are further divided by our modeled width categories (Figures I.5 through I.8). For each population, these tables provide estimates of the amount of currently available habitat of different types, the amount of habitat that has been cut off from anadromous fish passage, and the proportion of prime versus possible spawning habitat. Direct distance comparisons between these numbers and distances based on 1:100,000 stream measurements are not appropriate because of increased sinuosity and numbers of tributaries represented by our 1:24,000 stream measurements.

Both currently and historically, a large fraction of the available habitat that meets basic geomorphic criteria (e.g., gradient) is unsuitable for use because of issues of habitat quality, for example high water temperature, inadequate flow, or large deposits of fine sediments (Reeves 1995). The goal of this analysis is to estimate changes in habitat quantity over a very large area in a consistent manner. The true amount of usable habitat, including issues of habitat quality, may be only 40 to 60% of the habitat areas that meet the basic access and suitability criteria specified in our analysis. The percentage of potentially suitable habitat affected by habitat quality issues is likely not the same currently as it was historically. Our numbers are therefore likely to be overestimates of available habitat and underestimates of habitat loss. Until analyses are available that can include these more detailed issues, the results presented here provide a reasonable index of habitat change and numbers that are useful for making comparisons across watersheds and populations.

These data may be used to evaluate many other questions. For example, one might compare the kilometers of prime or possible habitat available to particular species within a given watershed (Table I.20). Such comparisons can aid in efforts to provide for multispecies recovery. Spatial identification of areas that are prime spawning or rearing habitat for multiple species could be used in prioritizing areas or barriers for restoration actions. Additional habitat criteria, such as width of riparian buffer will improve the usefulness of these estimates for answering additional questions.

Implied Fish Densities at Viability Thresholds

Ideally, we would like to estimate abundance viability criteria based on current and historical habitat capacity. However, as described above, the number of stream kilometers accessible to a particular population is highly uncertain. That uncertainty is compounded by a wide range of potential average and maximum fish densities in particular habitat types. Questions about the fraction of potentially suitable habitat made unsuitable by habitat quality issues further decrease our ability to estimate habitat capacity directly using currently available analysis tools. For example, we do not know the extent of thermal degradation or toxic contamination, nor do we know how these habitat impacts affect fish density in particular habitat

types. However, there remains a great need to relate abundance viability criteria to current, historical, and potential habitat quantities.

We use the habitat inventory to examine the implied fish densities of a range of abundance viability criteria (Tables I.21 through I.23). In this way, it will be possible to evaluate the feasibility of abundance viability criteria using current, historical, and potential habitat quantities, without diluting the habitat information with uncertainty about current or historical fish densities or capacities. Ideally, we would also like to assess the degree of habitat quality required to achieve the implied densities over current or potential habitat quantities. Fish densities estimated from our habitat inventory describe the average density required over a large area. Naturally, many habitat units that are suitable based on geomorphic criteria (e.g., gradient) would be unsuitable at a particular time because of issues of habitat quality (e.g., temperature, sediment composition, flow). This natural spatial variability in habitat quality increases required fish densities in the most suitable areas. Since our analyses describe the average fish density required, the more variability in the system (the more areas with low maximum fish densities), the higher the required fish densities in the best areas. Anthropogenic reductions in habitat quality (e.g., temperature, flow, sediment, toxics) would further increase fish densities required in the best remaining areas to meet population viability targets. Separate analyses identifying natural levels of habitat quality reductions, the spatial extent and degree of anthropogenic reductions in habitat quality, and effects of changes in habitat quality on fish densities and life-stage specific survivals would all be required to refine our current estimates.

Estimating the implied fish densities of abundance population viability criteria requires two pieces of information: the habitat area and the number of fish required to meet the abundance target. The implied fish density is simply the viability target divided by area. For comparison, we evaluate the density of current populations, as estimated by this approach, for each of four habitat areas: currently available prime, currently available possible, historically available prime, and historically available possible (Table I.22). Current abundance values are the average of the four most recent years of spawner counts. Some of the current abundance values may contain hatchery-origin spawners, as data did not always allow for distinction of natural- and hatchery-origin spawners.

We next evaluate five viability criteria across all four habitat area definitions: currently available prime, currently available possible, historically available prime, and historically available possible (Tables I.22 and I.23). Population definitions, current abundance estimates, and viability criteria under each of three scenarios are taken from Appendix E. Population criteria that we evaluated were developed from three scenarios that vary in their inclusion of hatchery fish and their projections of marine survival. Scenario 1 is based on extinction probabilities of declining to a four-year annual average of 50 spawners, calculated using population prediction intervals with 20 degrees of freedom for the variance estimate. The point estimate of the variance used to generate these targets is 0.05. The targets in scenario 1 assume that there are 0 hatchery-origin spawners present in any of the populations in the next 20 years. These targets also assume that the average of the marine survival index in the next 20 years is equal to long-term average marine survival (Table I.22). We evaluated two different extinction probabilities for this scenario (5% and 15%). Scenario 2 is identical, except that targets assume 5% of the spawners are of hatchery origin in every population over the next 20 years. The actual target size is still expressed in terms of natural-origin spawners. As in scenario 1, targets in scenario 2 assume that the average of the marine survival index in the next 20 years will be equal to the long-term average marine survival. Targets in scenario 3 assume that there are zero

hatchery-origin spawners present in any of the populations in the next 20 years (Table I.23), but scenario 3 targets assume that the average of the marine survival index in the next 20 years is 20% higher than the long-term average marine survival (Table I.23). Targets in scenario 4 assume both a hatchery influence and a change in marine survival. These targets assume that 5% of the spawners are of hatchery origin in every population over the next 20 years. Again, the target size is expressed only in terms of natural origin spawners. Scenario 4 targets also assume that the average of the marine survival index in the next 20 years is 20% higher than the long-term average marine survival (Table I.23). These scenarios are described in detail in Appendix D (Tables D.3 through D.6).

The implied fish densities presented here should be compared to ranges of species-specific fish densities from field observations, published literature, and/or historical records. A first step in evaluating this methodology will be to compare the estimates of current fish density to field observations. The fish densities under each of the viability criteria can be evaluated for their reasonableness: “Would it be reasonable to expect that we could observe average fish densities of this magnitude in this watershed?” If so, current habitat may be of sufficient quantity to support a population as large as the abundance criteria require. The habitat may not be of sufficient quality; this question should be addressed in separate analyses. The implied average fish density, given all historically available habitat, suggests whether the criteria are reasonable given all habitat that a population once used. If the implied average fish density over all historically available habitat is much higher than anything one might expect to see in the field, we have an indication that the criteria may be too high. As well as evaluating criteria, we can use the tables to examine the potential impact of reconnecting currently inaccessible habitat by comparing fish densities for currently and historically available habitats. Increasing the number of habitat quality predictors would make important refinements in these predictions. We have not divided the density estimates according to channel-width categories, but it would be possible to do so based on estimates of the proportion of spawning in large main stems, small main stems, large tributaries, and small tributaries for each species.

Conclusion

The broad-scale habitat inventory provides a method for making comparisons across and between ecoregions, watersheds, and ESUs. The first step of the analysis, a detailed inventory of habitat types, classified by accessibility, provides the building blocks for multiple recovery-related habitat analyses. Here it is used to estimate the implied fish densities of a range of abundance viability criteria. Implied densities from a range of future population scenarios can be evaluated with respect to species-specific fish densities from field observations, published literature, or historical records. There is uncertainty associated with our estimates of current and potential stream kilometers, as well as with our classification of prime versus possible habitat; we have identified and minimized these sources of uncertainty. In future analyses, we will also attempt to quantify the uncertainty. The inventory approach has a wide range of additional applications including estimating habitat quantities above individual passage barriers and developing models of in-stream habitat characteristics from landscape-scale digital data.

Table I.1 Reach-level channel attributes derived from drainage elevation models.

Landscape Derived/Modeled Stream Attributes	
Flow accumulation / drainage area	Steepest reach downstream (gradient)
Reach-averaged stream gradient	Left/right channel side-slope (~ 10 m)
Stream order (Strahler)	Left/right channel side-slope (~ 100 m)
Stream reach length	Left/right side valley floor width (m)
Mean annual precipitation depth (mm)	Lake or stream

Table I.2 Statistical models to predict stream width in 13 watersheds in the Willamette/Lower Columbia domain.

Watershed Name	Fourth-Field		AREA ^b	GRAD ^b	PRECIP ^b	BA*P ^c	R ^{2d}	Small Streams Excluded ^e
	HUC ^a	INT ^b						
Middle Columbia-Hood	70105	-1.39	0.52		0.0008		0.75	yes
Lower Columbia-Sandy	80001	0.65	0.42		0.0002		0.76	yes
Lower Columbia-Clatskanie	80003	1.58	0.25				0.15	
Lower Columbia-Middle Fork	80006	2.54	0.17		-0.0007		0.29	
Willamette	90001	1.72	0.15	-2.71			0.38	yes
Upper Willamette	90003	0.81	0.23		0.0002		0.30	
McKenzie	90004	0.12	0.67		0.0006	-0.0002	0.62	
North Santiam	90005	3.02	-0.73		-0.0006	0.0004	0.41	
South Santiam	90006	4.45	-0.90	-1.42	-0.0013	0.0006	0.71	
Middle Willamette	90007	1.07	0.39				0.50	
Tualatin	90010	0.58	0.31		0.0004		0.35	
Clackamas	90011	0.61	0.52				0.62	
Lower Willamette	90012	-0.98	0.29		0.0014		0.63	
All watersheds ^f		1.10	0.29		0.0001		0.41	

^a HUC = hydrologic unit codes

^b The columns intercept (INT), basin area (AREA), channel gradient (GRAD), and precipitation (PRECIP), describe potential predictor variables. If they are included in the final model for a particular watershed, the coefficient is presented in that column.

^c Basin Area * Precipitation (BA*P) describes an interaction term. If it was included in the final model, a coefficient is presented in that column. Basin area was log-transformed in all cases.

^d The R² value presented is the multiple R² value.

^e This column indicates whether streams within basin areas smaller than 1 km² were excluded from the analysis.

^f This row describes a model for all watersheds combined.

Table I.3 Barrier databases used to delineate accessible and inaccessible stream segments.

Data Set^a	Date Received	Source Date	Data Extent	No. Points (Not Always Unique)	Barrier Type	Passage-Related Information
Mthoodbarriers	06/01	6/1994	Mt. Hood National Forest, Oregon	124	Natural	No
Batemanbarriers	09/01	03/2000	Willamette Valley, coastal Oregon	635	Natural	Yes
BPA	10/01	10/2001	Washington, Oregon, Idaho, Montana	2,384 (326 in WLC)	Artificial	Limited
Wvndams	06/01	2000	Willamette Valley, coastal Oregon	213	Artificial	Yes
Wvncbars	10/01	02/2000	Willamette Valley, north coastal Oregon	709	Artificial	Yes
ODFW dams and fishways	07/01	1998	Oregon	744	Natural	Yes
Mvbdams	06/01	1995	Oregon, Washington, Idaho, Montana, Nevada	9,707 (1,030 in WLC)	Artificial	No
ORCulverts (three files)	05/01	1995	Western Oregon	4,267 (2,349 in WLC)	Artificial	Limited
WaBarriers	09/01	1999	Washington	3,365 (180 in WLC)	Artificial	Limited
NewWaBar	2002	03/2002 data recalled	SW Washington (WRIA ^b 24, 25, 26, 27, 28, 29)	2,011	Natural	Yes

^a Key to barriers databases:

Mthoodbarriers = Barriers in Mt. Hood National Forest (USFS 1994)

Batemanbarriers = Natural and man-made barriers to fish passage, western Oregon (Gresswell and Bateman 2000)

BPA = BPA dams and possible hydro sites (BPA 2001)

Wvndams = Western Oregon dams/barriers (StreamNet 2000)

Wvncbars = Western Oregon dams/barriers (StreamNet 2000)

ODFW dams and fishways = Nonspatial database from ODFW with fishway information (ODFW 2000)

Mvbdams = Interior Columbia Ecosystem Management Project (ICBEMP) from U.S. Army Corps of Engineers (Quigley et al. 2001)

ORCulverts = Culvert locations and passage info from Charlie Corrarino, ODFW Fish passage division (ODFW 2001b)

WaBarriers = Washington State man-made and natural barriers. Original data from Martin Hudson, WDFW; will be superseded by new Washington SSHIAP data (WDFW 1999).

NewWaBar = Washington barriers: New SSHIAP barrier data for southwest Washington (SSHIAP 2002)

^b WRIA = Water Resource Inventory Area

Table I.4 Gradient ranges for prime and possible habitat by species and life stage.^a

Species	Spawning	Rearing
<i>Possible</i>		
Chinook salmon	0.5–4%	0.5–3%
Steelhead	0.5–4% (summer)	0–7% (summer)
	0.5–6% (winter)	1.5–7% (winter)
Chum salmon	0–3.5%	
<i>Prime</i>		
Chinook salmon	1–2%	1–2%
Steelhead	3–4% (summer)	1–3% (summer)
	1–5% (winter)	1.5–7% (winter)

^a Based on interviews with Mark Wade, Oregon Department of Fish and Wildlife (ODFW), 90700 Fish Hatchery Road, Leaburg, OR 97489; Jeff Ziller, ODFW, Springfield Field Office, 3150 East Main St Springfield, OR 97478-5800; Gary Galovich, ODFW, South Willamette Watershed District Office, 7118 NE Vanderberg Avenue, Corvallis, OR 97330-9446; Kurt Schroeder and Ken Kenniston, ODFW, Corvallis Research Lab, 28655 Highway 34 Corvallis, OR 97333; Wayne Hunt, ODFW, Salem Field Office, 4412 Silverton Road NE Salem, OR 97305-2060; Steve Cramer, S.P. Cramer and Associates, Inc., 39330 Proctor Blvd., Sandy, OR 97055; Pat Connelly, U.S. Geological Survey, Columbia River Research Lab, 5501-A Cook-Underwood Rd., Cook, WA 98605; Joe Hymer, Washington Department of Fish and Wildlife, Region 5 Office, 2108 Grand Boulevard, Vancouver, WA 98661.

Table I.5 Accessible and inaccessible prime spawning kilometers for fall chinook populations in the Lower Columbia ESU by stream-width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Kalama	Main stem (sm)	11.77	0.00	2.28	0.00	0.00	0.00
	Tributary (lg)	7.44	0.20	2.51	0.00	0.00	0.00
	Tributary (sm)	5.06	0.39	3.13	0.04	0.00	0.00
Lewis Salmon	Main stem (lg)	0.86	0.33	0.00	0.00	0.00	0.00
	Main stem (sm)	13.52	8.12	1.07	2.27	3.69	0.00
	Tributary (lg)	26.38	10.92	1.76	8.25	4.71	0.00
	Tributary (sm)	46.47	30.10	1.73	24.70	18.93	0.67
Big Creek	Main stem (sm)	2.98	5.04	0.00	0.00	0.00	0.00
	Tributary (lg)	2.55	2.49	0.00	0.00	0.39	0.00
	Tributary (sm)	22.07	1.88	0.00	0.00	0.36	0.00
Big White Salmon	Main stem (lg)	0.00	5.29	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	6.20	0.00	0.00	0.00	0.00
	Tributary (sm)	0.00	11.57	0.00	0.00	0.00	0.00
Clackamas	Main stem (lg)	2.77	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	26.70	0.00	1.01	0.00	5.40	0.00
	Tributary (lg)	14.75	0.00	1.85	0.15	4.43	0.11
	Tributary (sm)	121.81	14.95	5.12	6.03	4.41	0.93
Clatskanie	Main stem (sm)	11.84	0.00	1.95	0.00	0.00	0.00
	Tributary (lg)	12.04	0.12	11.96	0.49	0.00	0.55
	Tributary (sm)	17.55	0.00	15.96	0.00	0.00	0.79
Coweeman	Main stem (sm)	11.34	0.00	0.25	0.00	0.00	0.00
	Tributary (lg)	4.75	1.98	2.18	0.00	0.00	0.00
	Tributary (sm)	3.33	0.56	1.53	0.07	0.00	0.00
Cowlitz	Main stem (lg)	1.28	2.25	0.00	0.00	0.00	0.00
	Main stem (sm)	14.67	32.89	5.71	0.00	1.37	0.00
	Tributary (lg)	36.69	42.50	18.62	0.00	4.48	0.00
	Tributary (sm)	70.39	75.43	18.47	0.00	8.66	0.67
Elochoman	Main stem (sm)	3.68	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	11.46	2.59	0.00	0.00	0.00	0.16
	Tributary (sm)	12.62	5.88	0.00	0.00	0.00	0.11
Grays	Main stem (sm)	7.16	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	16.49	0.00	1.48	0.00	0.00	0.00
	Tributary (sm)	21.62	0.00	3.83	0.00	0.00	0.00
Hood	Main stem (lg)	5.38	0.00	0.00	0.89	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	1.55	1.84	0.00
	Tributary (sm)	3.05	0.00	0.00	2.06	0.13	0.00
Lower gorge tributaries	Main stem (sm)	1.46	0.00	0.00	0.00	0.00	1.28
	Tributary (lg)	1.93	0.00	0.00	0.51	0.00	0.30
	Tributary (sm)	7.24	0.21	0.18	0.45	0.00	0.05
Mill	Main stem (sm)	15.09	0.00	1.73	0.00	0.00	0.00
	Tributary (lg)	11.32	0.00	3.91	0.00	0.00	0.00
	Tributary (sm)	10.73	1.44	5.38	0.00	0.00	0.11

Table 1.5 cont.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Sandy	Main stem (lg)	17.13	0.13	0.00	10.17	0.00	0.00
	Main stem (sm)	11.73	7.67	0.25	0.00	0.00	1.92
	Tributary (lg)	7.96	1.90	0.90	1.84	0.00	3.46
	Tributary (sm)	15.18	9.44	0.70	8.66	0.00	3.60
Scappoose	Main stem (lg)	0.06	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	4.93	0.00	0.00	0.00	0.31	0.00
	Tributary (lg)	13.53	3.13	3.71	0.00	8.53	0.00
	Tributary (sm)	34.31	7.98	4.65	2.71	7.12	0.12
Toutle	Main stem (lg)	8.97	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	18.54	9.18	1.11	0.36	1.23	0.63
	Tributary (lg)	12.44	7.21	7.14	1.99	0.04	1.89
	Tributary (sm)	21.27	14.41	6.67	6.59	0.05	1.07
Upper gorge tributaries	Main stem (lg)	0.00	0.00	0.00	0.00	0.30	0.00
	Main stem (sm)	0.84	0.99	5.82	0.00	0.13	0.13
	Tributary (lg)	0.79	0.45	1.80	0.00	0.00	0.00
	Tributary (sm)	4.73	0.00	6.25	0.00	0.00	0.00
Washougal	Main stem (lg)	2.48	0.00	0.00	1.28	0.43	0.00
	Main stem (sm)	0.94	1.86	0.52	4.01	3.14	0.00
	Tributary (lg)	1.14	8.78	0.50	2.17	2.63	0.00
	Tributary (sm)	2.62	14.11	0.31	2.50	1.55	0.00
Youngs	Main stem (sm)	5.67	5.57	5.84	0.00	0.50	0.00
	Tributary (lg)	12.48	0.21	6.85	0.00	0.00	0.82
	Tributary (sm)	38.11	0.38	1.74	0.00	0.11	0.54

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.6 Accessible and inaccessible possible spawning kilometers for fall chinook populations in the Lower Columbia ESU by stream width category.^a

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Kalama	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
Lewis Salmon	Main stem (lg)	2.53	2.42	0.00	0.00	0.00	0.00
	Main stem (sm)	36.66	30.24	3.36	8.17	8.46	0.00
	Tributary (lg)	74.13	33.04	7.75	19.13	19.73	0.00
	Tributary (sm)	149.13	94.52	9.68	68.26	51.88	1.59
Big Creek	Main stem (sm)	8.72	9.31	0.00	0.00	0.00	0.00
	Tributary (lg)	14.67	17.28	0.00	0.00	3.40	0.00
	Tributary (sm)	63.56	10.31	0.00	0.00	1.72	0.00
Big White Salmon	Main stem (lg)	0.33	12.69	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	11.56	0.00	0.00	0.00	0.00
	Tributary (sm)	0.00	46.35	0.00	0.00	0.00	0.00
Clackamas	Main stem (lg)	18.56	0.00	0.00	0.00	0.00	0.20
	Main stem (sm)	48.87	0.00	7.97	0.00	10.34	0.00
	Tributary (lg)	44.22	0.00	14.15	0.15	16.18	0.60
	Tributary (sm)	379.73	45.28	17.55	24.06	25.83	4.83
Clatskanie	Main stem (sm)	27.32	0.00	4.87	0.00	0.00	0.00
	Tributary (lg)	43.63	0.12	34.54	2.31	0.00	4.42
	Tributary (sm)	84.26	0.00	57.49	1.19	0.00	3.59
Coweeman	Main stem (sm)	27.16	0.00	0.95	0.00	0.00	0.00
	Tributary (lg)	17.98	6.75	6.79	0.00	0.00	0.00
	Tributary (sm)	15.96	3.06	6.48	0.19	0.00	0.00
Cowlitz	Main stem (lg)	4.94	12.37	0.00	0.00	0.00	0.00
	Main stem (sm)	35.10	107.40	44.81	0.00	4.67	0.00
	Tributary (lg)	100.66	134.61	68.93	0.00	12.90	0.00
	Tributary (sm)	229.32	246.44	65.89	0.00	30.47	1.68
Elochoman	Main stem (sm)	9.60	0.25	0.00	0.00	0.00	0.00
	Tributary (lg)	35.79	10.54	0.00	0.00	0.00	0.59
	Tributary (sm)	39.97	19.42	0.00	0.00	0.00	0.28
Grays	Main stem (sm)	21.44	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	46.38	0.00	5.01	0.00	0.00	0.00
	Tributary (sm)	64.70	0.00	10.28	0.00	0.00	0.00
Hood	Main stem (lg)	10.77	0.00	0.00	1.99	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	1.55	2.63	0.00
	Tributary (sm)	8.04	0.00	0.00	8.62	1.49	0.06
Lower gorge tributaries	Main stem (sm)	5.50	0.00	0.00	0.00	0.00	5.50
	Tributary (lg)	4.26	0.00	0.31	1.65	0.00	1.03
	Tributary (sm)	21.07	0.45	0.56	1.83	0.00	0.94
Mill	Main stem (sm)	26.67	0.00	3.65	0.00	0.00	0.00
	Tributary (lg)	43.17	0.36	13.33	0.00	0.00	0.00
	Tributary (sm)	47.61	4.98	18.80	0.00	0.00	0.66

Table 1.6 cont.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Sandy	Main stem (lg)	43.55	0.29	0.00	21.85	0.00	0.00
	Main stem (sm)	55.30	23.07	2.65	0.15	0.00	7.85
	Tributary (lg)	26.64	9.06	2.87	3.48	0.00	17.67
	Tributary (sm)	53.92	26.25	2.85	22.21	0.00	14.35
Scappoose	Main stem (lg)	0.61	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	7.24	0.00	0.00	0.00	0.78	0.00
	Tributary (lg)	33.88	7.59	9.89	0.00	16.85	0.00
	Tributary (sm)	105.93	27.44	17.26	7.62	24.93	0.70
Toutle	Main stem (lg)	20.78	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	49.66	23.26	4.15	0.81	2.22	1.98
	Tributary (lg)	38.01	23.72	26.15	6.35	0.98	6.12
	Tributary (sm)	75.16	49.09	27.72	22.73	0.26	4.92
Upper gorge tributaries	Main stem (lg)	0.87	0.00	0.00	0.00	1.09	0.00
	Main stem (sm)	2.31	2.89	16.47	0.00	0.37	1.87
	Tributary (lg)	1.02	1.13	6.31	0.00	0.00	0.37
	Tributary (sm)	17.01	0.25	24.35	0.00	0.00	0.00
Washougal	Main stem (lg)	5.78	0.00	0.00	2.17	2.06	0.00
	Main stem (sm)	3.81	8.05	0.97	9.46	8.98	0.00
	Tributary (lg)	2.28	22.00	2.38	8.98	10.94	0.00
	Tributary (sm)	11.90	49.70	1.00	9.25	8.46	0.00
Youngs	Main stem (sm)	18.54	11.55	15.37	0.00	1.64	0.00
	Tributary (lg)	35.94	0.60	20.61	0.00	0.29	3.89
	Tributary (sm)	121.68	4.43	17.20	0.00	0.28	2.63

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.7 Accessible and inaccessible prime spawning kilometers for winter steelhead populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Cispus	Main stem (lg)	0.00	3.10	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	38.84	61.72	0.00	0.00	0.00
	Tributary (lg)	0.00	2.98	70.74	0.00	0.00	0.00
	Tributary (sm)	0.00	15.08	69.28	0.00	0.00	0.00
Clackamas	Main stem (lg)	40.65	17.07	0.00	0.00	1.63	0.46
	Main stem (sm)	65.27	16.64	36.85	2.12	25.04	2.60
	Tributary (lg)	44.48	19.57	42.90	6.36	31.16	10.16
	Tributary (sm)	320.86	112.97	39.49	46.73	64.94	29.75
Coweeman	Main stem (sm)	21.03	0.00	0.85	0.00	0.00	0.00
	Tributary (lg)	19.37	7.31	6.88	0.00	0.00	0.00
	Tributary (sm)	19.28	4.69	7.57	0.39	0.00	0.00
East Fork	Main stem (sm)	14.52	0.00	3.70	0.00	9.98	0.00
Lewis	Tributary (lg)	42.79	7.43	4.33	5.86	16.18	0.00
	Tributary (sm)	68.15	53.07	4.74	5.69	10.65	1.01
Hood	Main stem (lg)	7.97	0.00	0.00	4.53	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	19.45	2.63	0.00
	Tributary (lg)	0.00	0.00	0.00	0.11	0.00	0.00
	Tributary (sm)	9.40	0.89	0.00	71.86	1.76	0.58
Kalama	Main stem (sm)	20.47	0.00	7.13	0.00	0.00	0.00
	Tributary (lg)	32.09	3.18	13.58	0.00	0.00	0.73
	Tributary (sm)	25.68	2.84	15.06	0.52	0.00	0.53
Lower Cowlitz	Main stem (lg)	1.71	0.30	0.00	0.00	0.00	0.00
	Main stem (sm)	22.50	3.28	1.94	0.00	0.35	0.00
	Tributary (lg)	74.82	21.09	7.93	0.00	8.43	0.00
	Tributary (sm)	234.64	60.46	4.24	0.00	35.53	1.90
Lewis	Main stem (lg)	0.86	1.62	0.00	0.00	0.00	0.00
	Main stem (sm)	6.27	104.50	0.00	0.00	0.00	0.00
	Tributary (lg)	19.59	96.29	2.07	0.00	2.34	0.00
	Tributary (sm)	40.59	93.87	8.34	0.00	10.21	0.00
Lower gorge tributaries	Main stem (sm)	5.33	0.00	0.14	0.00	0.00	6.38
	Tributary (lg)	4.49	0.00	0.40	1.53	0.00	1.63
	Tributary (sm)	19.02	0.21	0.47	2.17	0.00	1.82
North Fork Toutle	Main stem (lg)	9.32	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	17.44	17.51	2.90	0.36	2.50	1.42
	Tributary (lg)	22.47	20.72	21.78	6.64	0.58	4.48
	Tributary (sm)	59.73	48.53	27.58	23.64	0.44	3.82
Salmon	Main stem (sm)	0.87	0.49	0.00	3.25	2.97	0.00
	Tributary (lg)	3.05	1.57	1.15	10.47	1.89	0.00
	Tributary (sm)	45.36	20.15	1.17	59.94	26.88	0.66
Sandy	Main stem (lg)	19.11	0.34	0.00	10.47	0.00	0.00
	Main stem (sm)	62.55	28.20	2.65	0.62	0.00	10.25
	Tributary (lg)	29.78	10.81	4.15	4.17	0.00	21.39
	Tributary (sm)	63.30	27.04	3.69	25.42	0.00	19.68

Table 1.7 cont.

Population	Stream Size	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
South Fork	Main stem (sm)	19.35	0.00	0.00	0.00	0.00	0.00
Toutle	Tributary (lg)	16.58	3.11	6.36	0.00	0.43	0.77
	Tributary (sm)	23.89	4.75	6.24	0.00	0.00	0.73
Tilton	Main stem (sm)	0.00	18.89	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	26.62	1.58	0.00	0.00	0.00
	Tributary (sm)	0.00	40.77	0.39	0.00	0.00	0.00
Upper Cowlitz	Main stem (lg)	0.00	0.70	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	62.39	41.93	0.00	4.19	0.00
	Tributary (lg)	0.00	58.41	58.69	0.00	0.62	0.00
	Tributary (sm)	0.00	125.20	42.31	0.00	0.00	0.00
Upper gorge tributaries	Main stem (sm)	2.28	0.00	16.53	0.00	0.00	2.65
	Tributary (lg)	0.10	0.00	5.01	0.00	0.00	0.71
	Tributary (sm)	18.99	0.00	29.02	0.00	0.00	0.00
Washougal	Main stem (lg)	3.65	0.00	0.00	1.92	1.75	0.00
	Main stem (sm)	1.81	2.87	0.86	10.15	5.98	0.00
	Tributary (lg)	2.19	19.73	2.56	11.21	12.46	0.00
	Tributary (sm)	13.98	53.36	1.71	10.60	11.26	0.00
Wind	Main stem (lg)	0.87	0.00	0.00	0.00	9.02	5.52
	Main stem (sm)	0.00	2.75	0.00	0.00	8.16	7.28
	Tributary (lg)	1.08	1.47	1.72	0.00	4.55	7.00
	Tributary (sm)	0.29	0.44	1.41	0.00	13.86	12.61

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.8 Accessible and inaccessible possible spawning kilometers for winter steelhead populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Cispus	Main stem (lg)	0.00	9.03	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	52.94	70.16	0.00	0.00	0.00
	Tributary (lg)	0.00	5.64	89.54	0.00	0.00	0.00
	Tributary (sm)	0.00	19.22	94.10	0.00	0.00	0.00
Clackamas	Main stem (lg)	92.79	17.07	0.00	0.00	1.89	0.65
	Main stem (sm)	87.13	18.84	40.22	2.69	33.88	3.07
	Tributary (lg)	68.86	24.47	54.24	9.11	40.42	14.69
	Tributary (sm)	428.48	146.86	58.24	59.89	94.13	38.92
Coweeman	Main stem (sm)	28.45	0.00	0.95	0.00	0.00	0.00
	Tributary (lg)	25.90	8.94	9.39	0.00	0.00	0.00
	Tributary (sm)	30.59	7.22	10.20	0.45	0.00	0.00
East Fork	Main stem (sm)	26.14	0.00	4.28	0.00	11.72	0.00
Lewis	Tributary (lg)	55.77	8.53	6.39	8.51	21.04	0.00
	Tributary (sm)	89.82	67.64	7.18	8.23	17.32	1.36
Hood	Main stem (lg)	11.11	0.00	0.00	5.58	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	20.03	2.63	0.00
	Tributary (lg)	0.12	0.00	0.00	1.08	0.08	0.00
	Tributary (sm)	13.05	0.89	0.00	82.05	1.86	0.68
Kalama	Main stem (sm)	33.53	0.00	8.59	0.00	0.00	0.00
	Tributary (lg)	41.79	4.76	17.44	0.08	0.00	0.97
	Tributary (sm)	36.25	5.37	21.33	0.58	0.00	0.67
Lower Cowlitz	Main stem (lg)	4.94	0.39	0.00	0.00	0.00	0.00
	Main stem (sm)	35.76	3.69	1.94	0.00	1.34	0.00
	Tributary (lg)	116.13	37.79	10.34	0.00	14.05	0.00
	Tributary (sm)	324.14	89.88	6.77	0.00	45.71	2.33
Lewis	Main stem (lg)	2.53	2.80	0.00	0.00	0.00	0.00
	Main stem (sm)	10.84	143.74	0.00	0.00	0.00	0.00
	Tributary (lg)	28.30	132.35	2.74	0.00	5.06	0.00
	Tributary (sm)	54.84	130.45	11.77	0.00	13.88	0.00
Lower gorge tributaries	Main stem (sm)	7.04	0.00	0.32	0.00	0.00	8.28
	Tributary (lg)	5.12	0.00	0.69	2.05	0.00	1.74
	Tributary (sm)	28.97	0.45	0.87	2.81	0.00	2.57
North Fork Toutle	Main stem (lg)	20.78	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	27.24	26.57	4.52	0.81	2.85	2.06
	Tributary (lg)	28.77	26.92	28.96	8.81	0.73	5.81
	Tributary (sm)	83.77	67.21	40.02	34.93	0.75	6.32
Salmon	Main stem (sm)	2.14	1.70	0.00	8.33	5.62	0.00
	Tributary (lg)	5.35	1.57	1.72	15.41	4.92	0.00
	Tributary (sm)	61.98	26.96	1.41	81.78	36.52	0.66
Sandy	Main stem (lg)	44.15	0.82	0.00	21.85	0.00	0.00
	Main stem (sm)	70.86	35.55	2.85	1.00	0.00	13.42
	Tributary (lg)	39.27	15.90	5.29	4.70	0.00	27.91
	Tributary (sm)	83.26	38.73	5.03	30.02	0.00	26.43

Table 1.8 cont.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
South Fork	Main stem (sm)	26.34	0.00	0.00	0.00	0.00	0.00
Toutle	Tributary (lg)	23.09	4.11	8.59	0.00	0.77	1.42
	Tributary (sm)	31.37	6.44	8.69	0.00	0.00	1.23
Tilton	Main stem (sm)	0.00	27.05	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	38.02	2.35	0.00	0.00	0.00
	Tributary (sm)	0.00	55.13	0.55	0.00	0.00	0.00
Upper Cowlitz	Main stem (lg)	0.00	3.04	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	80.91	49.38	0.00	4.71	0.00
	Tributary (lg)	0.00	86.65	76.37	0.00	1.14	0.00
	Tributary (sm)	0.00	181.81	64.80	0.00	0.00	0.00
Upper gorge tributaries	Main stem (sm)	2.42	0.00	23.24	0.00	0.00	3.84
	Tributary (lg)	0.10	0.00	6.75	0.00	0.00	0.82
	Tributary (sm)	28.56	0.00	36.73	0.00	0.00	0.00
Washougal	Main stem (lg)	6.27	0.00	0.00	2.29	2.06	0.00
	Main stem (sm)	4.10	8.44	1.06	12.59	9.92	0.00
	Tributary (lg)	2.53	27.24	3.82	14.59	16.13	0.00
	Tributary (sm)	20.97	73.55	1.99	14.79	16.12	0.00
Wind	Main stem (lg)	0.87	0.00	0.00	0.00	10.72	8.73
	Main stem (sm)	0.00	2.89	0.00	0.00	11.13	8.97
	Tributary (lg)	1.30	1.63	1.86	0.00	5.60	8.48
	Tributary (sm)	0.39	0.77	1.70	0.00	18.89	16.49

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.9 Accessible and inaccessible potential spawning kilometers for chum populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Big Creek	Main stem (sm)	12.95	9.77	0.00	0.00	0.00	0.00
	Tributary (lg)	22.16	14.85	0.00	0.00	4.69	0.00
	Tributary (sm)	71.72	9.14	0.00	0.00	1.89	0.00
Clackamas	Main stem (lg)	36.96	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	0.03	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	43.75	0.00	0.00	0.00	1.31	0.00
	Tributary (sm)	325.27	55.17	0.00	0.00	8.49	0.00
Clatskanie	Main stem (sm)	56.51	0.00	13.22	0.00	0.00	0.00
	Tributary (lg)	71.84	0.12	43.43	1.71	0.00	3.33
	Tributary (sm)	111.12	0.00	53.03	0.67	0.00	2.46
Cowlitz	Main stem (lg)	120.58	3.61	0.00	0.00	0.00	0.00
	Main stem (sm)	131.41	6.56	1.94	0.00	4.67	0.00
	Tributary (lg)	210.73	62.55	17.79	0.00	18.84	0.00
	Tributary (sm)	371.77	87.42	19.37	1.47	28.44	2.18
Elochoman	Main stem (sm)	33.64	0.66	0.00	0.00	0.00	0.00
	Tributary (lg)	55.16	13.99	0.00	0.00	0.00	0.95
	Tributary (sm)	51.02	23.02	0.00	0.00	0.00	0.11
Grays	Main stem (sm)	45.84	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	74.00	0.00	6.80	0.00	0.00	0.00
	Tributary (sm)	109.11	0.70	14.15	0.00	0.00	0.00
Kalama	Main stem (sm)	61.28	0.00	4.99	0.00	0.00	0.00
	Tributary (lg)	25.96	1.67	16.52	0.00	0.00	0.25
	Tributary (sm)	26.22	4.83	12.53	0.12	0.00	0.20
Lewis	Main stem (lg)	30.39	38.90	0.00	0.00	3.67	0.00
	Main stem (sm)	75.56	42.62	3.14	0.00	5.26	0.00
	Tributary (lg)	88.43	36.63	7.04	5.65	20.98	0.00
	Tributary (sm)	124.66	79.31	7.59	4.46	22.81	1.04
Lower gorge tributaries	Main stem (lg)	0.00	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	14.17	0.23	0.00	0.00	0.00	4.90
	Tributary (lg)	12.75	0.00	0.31	1.62	0.00	0.99
	Tributary (sm)	51.13	0.45	0.51	1.67	0.00	0.96
Millcreek	Main stem (lg)	2.51	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	39.22	0.00	5.39	0.00	0.00	0.00
	Tributary (lg)	55.50	0.22	12.95	0.00	0.00	0.07
	Tributary (sm)	63.10	4.35	17.22	0.00	0.00	3.90
Salmon Creek	Main stem (lg)	0.00	0.86	0.00	0.00	10.78	0.00
	Main stem (sm)	10.22	10.09	0.00	21.79	15.98	0.00
	Tributary (lg)	19.13	1.47	1.04	27.98	17.66	0.00
	Tributary (sm)	77.21	30.23	0.41	72.87	45.82	0.58
Sandy	Main stem (lg)	50.33	2.01	0.00	17.63	0.00	0.00
	Main stem (sm)	8.56	17.38	0.00	0.00	0.00	5.68
	Tributary (lg)	16.80	4.42	2.23	1.21	0.00	14.86
	Tributary (sm)	38.28	22.61	2.46	15.15	0.00	10.34

Table 1.9 cont.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Scappoose	Main stem (lg)	37.17	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	29.19	0.00	0.00	0.00	24.86	0.00
	Tributary (lg)	57.64	10.26	14.77	0.00	9.45	0.00
	Tributary (sm)	173.84	30.05	15.01	9.81	27.47	0.65
Upper gorge tributaries	Main stem (lg)	8.95	8.48	0.00	0.00	2.79	0.00
	Main stem (sm)	4.79	2.60	21.70	0.00	0.37	2.01
	Tributary (lg)	1.37	0.85	5.89	0.00	0.00	0.27
	Tributary (sm)	36.37	10.61	22.16	0.00	0.05	0.00
Washougal	Main stem (lg)	18.10	0.00	0.00	8.38	6.56	0.00
	Main stem (sm)	7.88	27.75	1.37	11.90	13.90	0.00
	Tributary (lg)	4.17	31.91	2.91	9.04	10.14	0.00
	Tributary (sm)	16.73	55.97	0.54	9.90	8.10	0.00
Youngs	Main stem (sm)	45.78	13.47	20.08	0.00	1.64	0.00
	Tributary (lg)	67.58	0.60	19.88	0.00	0.29	5.68
	Tributary (sm)	153.89	3.50	13.69	0.00	0.28	6.46

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.10 Accessible and inaccessible prime spawning kilometers for late fall chinook populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
East Fork	Main stem (sm)	8.94	0.00	1.07	0.00	0.72	0.00
Lewis	Tributary (lg)	16.43	2.56	1.24	2.06	2.44	0.00
	Tributary (sm)	19.10	15.56	0.62	1.25	2.78	0.31
North Fork	Main stem (lg)	0.86	0.33	0.00	0.00	0.00	0.00
Lewis	Main stem (sm)	3.98	7.63	0.00	0.00	0.00	0.00
	Tributary (lg)	8.00	7.74	0.39	0.00	0.56	0.00
	Tributary (sm)	12.19	6.05	1.11	0.00	2.87	0.00
Sandy	Main stem (lg)	13.10	0.00	0.00	10.17	0.00	0.00
	Main stem (sm)	11.57	0.00	0.25	0.00	0.00	1.92
	Tributary (lg)	7.96	0.00	0.00	1.84	0.00	3.46
	Tributary (sm)	15.18	3.66	0.00	8.66	0.00	3.60

^a Mainstem (lg) > 25 m; mainstem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.11 Accessible and inaccessible possible spawning kilometers for late fall chinook populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
East Fork Lewis	Main stem (sm)	23.96	0.00	3.36	0.00	2.85	0.00
	Tributary (lg)	46.37	7.33	4.90	5.05	11.22	0.00
	Tributary (sm)	63.49	47.60	3.07	4.91	10.12	0.93
North Fork	Main stem (lg)	2.53	2.42	0.00	0.00	0.00	0.00
Lewis	Main stem (sm)	10.84	28.54	0.00	0.00	0.00	0.00
	Tributary (lg)	22.67	24.32	1.81	0.00	3.58	0.00
	Tributary (sm)	37.42	23.88	6.04	0.00	9.45	0.00
Sandy	Main stem (lg)	37.81	0.00	0.00	21.85	0.00	0.00
	Main stem (sm)	52.20	0.00	2.65	0.15	0.00	7.85
	Tributary (lg)	26.64	0.00	0.49	3.48	0.00	17.67
	Tributary (sm)	52.87	7.98	0.22	22.21	0.00	14.35

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.12 Accessible and inaccessible prime spawning kilometers for summer steelhead populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
East Fork	Main stem (sm)	1.33	0.00	1.69	0.00	1.79	0.00
Lewis	Tributary (lg)	8.52	1.81	1.00	0.90	4.24	0.00
	Tributary (sm)	15.89	11.82	1.27	1.55	1.77	0.27
Hood	Main stem (lg)	0.76	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	7.22	3.36	0.00
	Tributary (lg)	0.00	0.76	0.00	0.00	3.71	0.55
	Tributary (sm)	1.36	0.00	0.00	5.40	3.35	0.47
Kalama	Main stem (sm)	1.42	0.00	1.80	0.00	0.00	0.00
	Tributary (lg)	7.46	1.20	4.42	0.00	0.00	0.20
	Tributary (sm)	6.69	0.80	4.26	0.14	0.00	0.45
Lewis	Main stem (sm)	0.22	21.83	0.00	0.00	0.00	0.00
	Tributary (lg)	3.44	26.07	0.72	0.00	0.66	0.00
	Tributary (sm)	9.21	27.12	2.46	0.00	2.71	0.00
Washougal	Main stem (lg)	0.00	0.00	0.00	0.21	0.44	0.00
	Main stem (sm)	0.44	0.00	0.33	1.66	0.83	0.00
	Tributary (lg)	0.48	3.18	0.46	2.97	2.97	0.00
	Tributary (sm)	4.10	13.10	0.53	2.78	3.28	0.00
Wind	Main stem (lg)	0.24	0.00	0.00	0.00	0.91	0.00
	Main stem (sm)	0.00	0.37	0.00	0.00	0.59	0.25
	Tributary (lg)	0.00	0.30	0.18	0.00	1.44	2.16
	Tributary (sm)	0.00	0.08	0.36	0.00	4.20	3.15

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.13 Accessible and inaccessible possible spawning kilometers for summer steelhead populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
East Fork	Main stem (sm)	23.96	0.00	3.36	0.00	9.47	0.00
Lewis	Tributary (lg)	46.37	7.33	4.90	6.15	16.09	0.00
	Tributary (sm)	63.49	47.60	3.07	4.91	10.31	0.93
Hood	Main stem (lg)	10.77	0.00	0.00	1.99	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	14.14	35.62	0.00
	Tributary (lg)	0.00	2.33	0.00	0.00	6.18	1.39
	Tributary (sm)	8.04	0.27	0.00	13.65	6.77	0.90
Kalama	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
Lewis	Main stem (lg)	2.53	2.80	0.00	0.00	0.00	0.00
	Main stem (sm)	10.84	121.30	0.00	0.00	0.00	0.00
	Tributary (lg)	22.67	77.42	1.81	0.00	3.58	0.00
	Tributary (sm)	37.42	76.42	6.04	0.00	9.45	0.00
Washougal	Main stem (lg)	5.78	0.00	0.00	2.17	2.06	0.00
	Main stem (sm)	3.81	8.05	0.97	9.46	8.98	0.00
	Tributary (lg)	2.28	22.00	2.38	8.98	10.94	0.00
	Tributary (sm)	11.90	49.70	1.00	9.25	8.46	0.00
Wind	Main stem (lg)	0.87	0.00	0.00	0.00	9.35	8.73
	Main stem (sm)	0.00	2.89	0.00	0.00	9.22	8.67
	Tributary (lg)	1.02	1.13	1.86	0.00	3.56	5.46
	Tributary (sm)	0.29	0.25	0.94	0.00	12.07	10.84

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.14 Accessible and inaccessible prime spawning kilometers for spring chinook populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Big White Salmon	Main stem (lg)	0.00	10.76	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	13.52	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	12.70	0.00	0.00	0.00	0.00
	Tributary (sm)	0.00	41.91	0.00	0.00	0.00	0.00
Cispus	Main stem (lg)	0.00	2.13	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	22.70	17.19	0.00	0.00	0.00
	Tributary (lg)	0.00	0.94	15.74	0.00	0.00	0.00
	Tributary (sm)	0.00	2.99	14.49	0.00	0.00	0.00
Hood	Main stem (lg)	5.38	0.00	0.00	4.37	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	7.95	16.47	0.00
	Tributary (lg)	0.00	0.00	0.00	0.00	0.57	0.00
	Tributary (sm)	3.05	0.53	0.00	12.80	0.99	0.16
Kalama	Main stem (sm)	11.77	0.00	2.28	0.00	0.00	0.00
	Tributary (lg)	7.44	0.20	2.51	0.00	0.00	0.00
	Tributary (sm)	5.06	0.39	3.13	0.04	0.00	0.00
Lewis	Main stem (lg)	0.86	0.55	0.00	0.00	0.00	0.00
	Main stem (sm)	3.98	39.23	0.00	0.00	0.00	0.00
	Tributary (lg)	8.00	20.50	0.39	0.00	0.56	0.00
	Tributary (sm)	12.29	19.82	1.11	0.00	2.87	0.00
Sandy	Main stem (lg)	16.28	0.13	0.00	10.17	0.00	0.00
	Main stem (sm)	10.57	7.67	0.25	0.00	0.00	1.92
	Tributary (lg)	4.58	1.90	0.90	1.84	0.00	1.58
	Tributary (sm)	4.89	5.90	0.70	7.28	0.00	2.68
Tilton	Main stem (sm)	0.00	8.61	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	9.01	0.08	0.00	0.00	0.00
	Tributary (sm)	0.00	13.53	0.00	0.00	0.00	0.00
Toutle	Main stem (lg)	8.97	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	18.54	9.18	1.11	0.36	1.23	0.63
	Tributary (lg)	12.44	7.21	7.14	1.99	0.04	1.89
	Tributary (sm)	21.27	14.47	6.67	6.59	0.05	1.07
Upper Cowlitz	Main stem (sm)	0.00	22.06	4.06	0.00	1.37	0.00
	Tributary (lg)	0.00	20.67	11.52	0.00	0.00	0.00
	Tributary (sm)	0.00	41.41	12.02	0.00	0.00	0.00

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.15 Accessible and inaccessible possible spawning kilometers for spring chinook populations in the Lower Columbia ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Big White Salmon	Main stem (lg)	0.33	26.09	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	35.82	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	26.62	0.00	0.00	0.00	0.00
	Tributary (sm)	0.00	142.99	0.00	0.00	0.00	0.00
Cispus	Main stem (lg)	0.00	9.03	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	47.83	58.60	0.00	0.00	0.00
	Tributary (lg)	0.00	4.74	63.72	0.00	0.00	0.00
	Tributary (sm)	0.00	14.09	58.22	0.00	0.00	0.00
Hood	Main stem (lg)	10.77	0.00	0.00	5.58	0.00	0.00
	Main stem (sm)	0.00	0.00	0.00	18.70	35.62	0.00
	Tributary (lg)	0.00	0.00	0.00	0.00	6.18	1.39
	Tributary (sm)	8.04	0.89	0.00	57.90	6.77	1.36
Kalama	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
Lewis	Main stem (lg)	2.53	2.80	0.00	0.00	0.00	0.00
	Main stem (sm)	10.84	121.30	0.00	0.00	0.00	0.00
	Tributary (lg)	22.67	77.42	1.81	0.00	3.58	0.00
	Tributary (sm)	37.58	76.42	6.04	0.00	9.45	0.00
Sandy	Main stem (lg)	37.13	0.29	0.00	21.85	0.00	0.00
	Main stem (sm)	48.51	23.07	2.65	0.15	0.00	7.85
	Tributary (lg)	17.00	9.06	2.87	3.48	0.00	11.63
	Tributary (sm)	21.55	18.66	2.85	17.05	0.00	11.48
Tilton	Main stem (sm)	0.00	24.98	0.00	0.00	0.00	0.00
	Tributary (lg)	0.00	27.61	1.10	0.00	0.00	0.00
	Tributary (sm)	0.00	40.02	0.26	0.00	0.00	0.00
Toutle	Main stem (lg)	20.78	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	49.66	23.26	4.15	0.81	2.22	1.98
	Tributary (lg)	38.01	23.72	26.15	6.35	0.98	6.12
	Tributary (sm)	75.16	49.15	27.72	22.73	0.26	4.92
Upper Cowlitz	Main stem (lg)	0.00	3.04	0.00	0.00	0.00	0.00
	Main stem (sm)	0.00	69.17	34.99	0.00	3.68	0.00
	Tributary (lg)	0.00	66.99	45.47	0.00	0.00	0.00
	Tributary (sm)	0.00	132.93	42.91	0.00	0.00	0.00

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.16 Accessible and inaccessible prime spawning kilometers for spring chinook populations in the Willamette ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Calapooia	Main stem (lg)	0.71	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	12.92	0.91	0.00	0.00	0.00	0.00
	Tributary (lg)	8.45	9.03	0.06	5.02	0.00	0.00
	Tributary (sm)	14.33	17.48	0.00	11.61	0.00	0.00
Clackamas	Main stem (lg)	33.71	8.30	0.00	0.00	0.29	0.00
	Main stem (sm)	13.73	4.20	7.19	0.00	10.24	0.45
	Tributary (lg)	5.80	7.29	6.03	0.00	7.91	3.89
	Tributary (sm)	26.75	14.28	6.66	1.77	13.45	5.09
McKenzie	Main stem (lg)	0.49	0.00	0.00	1.84	0.00	0.00
	Main stem (sm)	10.73	7.19	6.68	15.54	0.00	5.36
	Tributary (lg)	12.14	9.45	19.99	6.79	0.74	1.84
	Tributary (sm)	20.07	10.70	22.06	15.69	0.00	2.82
Middle Fork Willamette	Main stem (lg)	0.77	9.22	0.00	0.00	0.00	0.00
	Main stem (sm)	9.15	29.30	1.03	6.55	9.98	1.95
	Tributary (lg)	2.02	14.92	0.76	3.15	7.23	1.00
	Tributary (sm)	8.30	25.39	0.41	9.10	1.63	1.15
Molalla	Main stem (sm)	51.94	0.00	2.79	2.79	13.79	0.00
	Tributary (lg)	17.00	21.88	5.03	3.43	2.04	0.50
	Tributary (sm)	49.81	62.14	3.44	6.77	2.19	2.51
North Santiam	Main stem (sm)	0.62	6.93	6.19	5.56	0.50	0.00
	Tributary (lg)	2.41	8.78	5.43	8.13	1.67	0.00
	Tributary (sm)	10.51	11.30	2.59	18.81	0.13	0.04
South Santiam	Main stem (lg)	0.37	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	19.91	6.69	2.15	3.74	6.77	2.35
	Tributary (lg)	22.73	7.99	11.67	2.40	7.59	1.60
	Tributary (sm)	65.97	46.66	1.46	15.78	1.45	1.67

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.17 Accessible and inaccessible possible spawning kilometers for spring chinook populations in the Willamette ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Calapooia	Main stem (lg)	4.12	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	28.13	3.19	0.00	0.47	0.00	0.00
	Tributary (lg)	26.07	24.25	0.34	13.31	0.00	0.00
	Tributary (sm)	54.37	61.67	0.18	36.98	0.00	0.00
Clackamas	Main stem (lg)	88.91	16.62	0.00	0.00	1.40	0.65
	Main stem (sm)	42.58	15.17	23.77	0.00	30.26	2.23
	Tributary (lg)	18.37	19.71	23.34	0.00	29.57	10.26
	Tributary (sm)	94.90	54.36	26.46	9.01	54.45	21.57
McKenzie	Main stem (lg)	4.11	0.00	0.00	5.88	0.00	0.00
	Main stem (sm)	24.78	34.61	29.39	55.93	0.00	11.94
	Tributary (lg)	45.15	27.58	65.41	21.27	4.85	7.70
	Tributary (sm)	72.00	36.83	85.33	49.28	0.13	10.60
Middle Fork Willamette	Main stem (lg)	4.91	20.57	0.00	0.00	0.00	0.00
	Main stem (sm)	22.49	83.17	5.04	27.26	25.79	10.00
	Tributary (lg)	11.53	47.15	3.87	8.61	24.76	3.13
	Tributary (sm)	32.09	77.67	1.98	28.73	10.44	3.52
Molalla	Main stem (lg)	1.29	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	125.07	0.26	9.45	3.31	32.41	0.00
	Tributary (lg)	67.82	73.74	29.58	7.23	14.60	2.55
	Tributary (sm)	151.08	182.14	14.88	22.57	6.75	8.12
North Santiam	Main stem (lg)	0.17	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	3.35	27.13	22.31	24.83	2.55	0.00
	Tributary (lg)	11.80	27.97	20.45	23.02	6.95	0.28
	Tributary (sm)	37.48	41.14	14.08	61.41	1.00	0.24
South Santiam	Main stem (lg)	2.81	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	57.39	24.42	8.98	6.76	20.66	6.60
	Tributary (lg)	61.13	34.80	37.31	7.03	27.09	13.79
	Tributary (sm)	212.11	153.29	10.12	44.08	6.22	5.49

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.18 Accessible and inaccessible prime spawning kilometers for winter steelhead populations in the Willamette ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Calapooia	Main stem (lg)	0.71	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	17.98	0.91	0.00	0.00	0.00	0.00
	Tributary (lg)	19.67	21.03	0.40	11.02	0.00	0.00
	Tributary (sm)	52.32	57.55	0.18	37.05	0.00	0.00
Coast Range	Main stem (lg)	0.38	0.00	0.00	0.09	0.00	0.00
	Main stem (sm)	11.96	4.27	11.19	15.95	5.24	3.08
	Tributary (lg)	45.28	71.92	26.49	132.90	13.85	12.62
	Tributary (sm)	236.02	408.45	30.66	474.95	14.09	35.95
Molalla	Main stem (sm)	76.06	0.00	11.07	3.09	30.96	0.00
	Tributary (lg)	53.55	37.05	37.47	4.49	25.59	2.39
	Tributary (sm)	131.77	152.46	19.95	23.54	9.68	8.43
North Santiam	Main stem (sm)	1.51	20.13	20.58	10.70	2.37	0.00
	Tributary (lg)	7.19	30.00	24.49	20.19	6.88	0.37
	Tributary (sm)	23.47	48.27	19.25	55.58	1.54	0.29
South Santiam	Main stem (lg)	0.37	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	28.68	18.71	6.77	3.74	19.73	6.56
	Tributary (lg)	41.83	28.88	45.58	7.79	31.32	19.13
	Tributary (sm)	194.84	135.88	14.14	49.18	8.36	7.57

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.19 Accessible and inaccessible possible spawning kilometers for winter steelhead populations in the Willamette ESU by stream width category.

Population	Stream Size^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
Calapooia	Main stem (lg)	4.12	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	29.29	3.19	0.00	0.47	0.00	0.00
	Tributary (lg)	29.77	26.96	0.51	14.16	0.00	0.00
	Tributary (sm)	75.39	84.57	0.43	50.14	0.00	0.00
Coast Range	Main stem (lg)	2.27	0.00	0.00	3.80	0.00	0.00
	Main stem (sm)	41.71	4.89	13.97	46.54	9.03	8.31
	Tributary (lg)	76.67	117.93	34.38	191.89	15.56	20.12
	Tributary (sm)	321.55	554.15	41.17	650.07	17.25	52.88
Molalla	Main stem (lg)	1.29	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	130.57	0.26	11.27	3.31	35.41	0.00
	Tributary (lg)	80.47	79.27	52.46	7.55	35.82	4.93
	Tributary (sm)	185.49	223.68	28.66	31.07	13.06	9.81
North Santiam	Main stem (lg)	0.17	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	3.72	30.28	27.92	25.72	4.01	0.00
	Tributary (lg)	13.42	44.15	34.05	28.22	8.95	0.37
	Tributary (sm)	42.67	62.32	27.50	80.90	2.43	0.51
South Santiam	Main stem (lg)	2.81	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	61.51	27.72	11.26	6.76	23.96	6.60
	Tributary (lg)	66.90	49.43	63.19	12.23	41.52	25.19
	Tributary (sm)	282.58	198.83	19.36	68.98	13.25	10.96

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.20 A comparison of accessible and inaccessible prime spawning kilometers for multiple species of concern in the Kalama and Clackamas watersheds.

Population	Stream Size ^a	Accessible	Inaccessible Due to Man-made Barriers	Inaccessible Due to Natural Barriers	Partially Accessible Due to Man-made Barriers	Partially Accessible Due to Natural Barriers	Unknown
KALAMA							
Fall chinook	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
Winter steelhead	Main stem (sm)	33.53	0.00	8.59	0.00	0.00	0.00
	Tributary (lg)	41.79	4.76	17.44	0.08	0.00	0.97
	Tributary (sm)	36.25	5.37	21.33	0.58	0.00	0.67
Chum	Main stem (sm)	61.28	0.00	4.99	0.00	0.00	0.00
	Tributary (lg)	25.96	1.67	16.52	0.00	0.00	0.25
	Tributary (sm)	26.22	4.83	12.53	0.12	0.00	0.20
Summer steelhead	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
Spring chinook	Main stem (sm)	30.39	0.00	6.04	0.00	0.00	0.00
	Tributary (lg)	26.26	2.34	11.58	0.00	0.00	0.39
	Tributary (sm)	20.91	2.76	11.00	0.18	0.00	0.45
CLACKAMAS							
Fall chinook	Main stem (lg)	18.56	0.00	0.00	0.00	0.00	0.20
	Main stem (sm)	48.87	0.00	7.97	0.00	10.34	0.00
	Tributary (lg)	44.22	0.00	14.15	0.15	16.18	0.60
	Tributary (sm)	379.73	45.28	17.55	24.06	25.83	4.83
Winter steelhead	Main stem (lg)	92.79	17.07	0.00	0.00	1.89	0.65
	Main stem (sm)	87.13	18.84	40.22	2.69	33.88	3.07
	Tributary (lg)	68.86	24.47	54.24	9.11	40.42	14.69
	Tributary (sm)	428.48	146.86	58.24	59.89	94.13	38.92
Chum	Main stem (lg)	36.96	0.00	0.00	0.00	0.00	0.00
	Main stem (sm)	0.03	0.00	0.00	0.00	0.00	0.00
	Tributary (lg)	43.75	0.00	0.00	0.00	1.31	0.00
	Tributary (sm)	325.27	55.17	0.00	0.00	8.49	0.00
Spring chinook	Main stem (lg)	88.91	16.62	0.00	0.00	1.40	0.65
	Main stem (sm)	42.58	15.17	23.77	0.00	30.26	2.23
	Tributary (lg)	18.37	19.71	23.34	0.00	29.57	10.26
	Tributary (sm)	94.90	54.36	26.46	9.01	54.45	21.57

^a Main stem (lg) > 25 m; main stem (sm) 10–25 m; tributary (lg) 5–10 m; tributary (sm) < 5 m.

Table I.21 Currently and historically available prime and possible spawning kilometers for those populations with estimated viability targets.
Estimated population viability targets for each of four scenarios are provided in the final columns.

ESU	Population ^a	Possible Current	Possible Historical	Prime Current	Prime Historical	Current Abundance	Scenario 1 ^b (5%)	Scenario 1 ^b (15%)	Scenario 2 ^c (5%)	Scenario 3 ^d (5%)	Scenario 4 ^e (5%)
Columbia chum	Grays (winter)	228.95	229.65			960	3,300	2,000	6,300	3,900	7,100
	Lower gorge (winter)	81.33	82.02			375	1,600	1,000	3,000	1,900	3,500
Upper Willamette steelhead	North Santiam (winter)	210.23	346.97	129.43	227.82	1,382	4,500	2,600	8,700	5,200	9,900
	South Santiam (winter)	580.51	856.49	385.86	569.33	916	3,300	1,900	6,000	3,800	7,100
	Mollala (winter)	524.04	827.25	358.74	548.24	655	2,400	1,500	4,700	2,900	5,500
	Calapooia (winter)	203.33	318.06	138.75	218.25	104	700	400	1,100	700	1,300
Upper Willamette chinook	McKenzie (spring)	283.38	382.41	84.03	111.38	1,861	5,700	3,300	10,700	6,600	13,000
	Clackamas (spring)	369.47	475.33	113.64	147.71	1,103	3,600	2,200	7,000	4,300	8,300
Lower Columbia steelhead	Wind (winter)	48.9	54.18	37.83	42.49	286	1,300	800	2,400	1,500	2,900
	South Fork Toutle (summer)	81.57	92.12	60.25	68.1	463	1,900	1,100	3,600	2,200	4,100
	Sandy (winter)	295.09	386.1	215.42	281.81	965	3,300	2,000	6,400	3,800	7,400
	North Fork Toutle (winter)	209.43	330.13	143.12	229.88	176	900	600	1,700	1,100	1,900
	Kalama (winter)	112.23	122.36	78.76	84.78	539	2,200	1,300	4,000	2,300	4,600
	Kalama (summer)	77.74	82.84	15.71	17.71	443	1,800	1,100	3,400	2,100	4,000
	Hood (winter)	137.58	138.47	117.71	118.6	593	2,300	1,400	4,400	2,700	5,000
	Hood (summer)	97.16	99.77	25.16	25.93	560	2,100	1,300	4,200	2,500	4,800
	Clackamas (winter)	919.27	1126.52	649.23	815.49	386	1,600	1,000	3,100	1,900	3,500
Lower Columbia chinook	White Salmon (fall)	0.33	70.94	0	23.07	163	900	600	1,600	1,000	1,800
	Washougal (fall)	84.05	163.8	24.91	49.66	735	2,700	1,600	5,200	3,000	5,800
	Sandy (late fall)	217.21	225.19	68.48	72.14	1,095	3,600	2,200	7,000	4,300	8,400
	North Fork Lewis (bright)	86.66	364.61	28.56	108.65	8,915	20,300	12,000	39,300	23,800	47,900
	Mill, Abernathy, Germ. (fall)	117.45	122.79	37.15	38.59	348	1,500	1,000	2,900	1,700	3,300
	Kalama (fall)	77.74	82.84	24.31	24.9	1,192	3,900	2,400	7,600	4,400	8,700
	Grays (fall)	132.52	132.52	45.26	45.26	62	500	300	800	500	900
	Elochoman	85.36	115.57	27.76	36.23	297	1,400	800	2,500	1,500	2,800
	Cowlitz (fall)	418.05	918.86	137.54	290.61	748	2,800	1,600	5,300	3,200	6,100
	Coweeman (fall)	61.28	71.09	19.49	22.02	425	1,800	1,100	3,400	2,000	3,800

Clackamas (fall)	567.95	613.22	186.45	201.4	164	900	600	1,600	1,000	1,800
Sandy (fall)	227.1	285.77	72.67	91.8	140	800	500	1,400	900	1,700

^a Populations and scenarios are from Appendix D (Tables D.3–D.6).

^b The two targets for scenario 1 represent a 5 and a 15% chance of extinction in the next 100 years. Scenario 1 assumes no hatchery influence and no change in marine survival.

^c Scenario 2 assumes some hatchery influence and no change in marine survival.

^d Scenario 3 assumes no hatchery influence and a change in marine survival.

^e Scenario 4 assumes both a hatchery influence and a change in marine survival.

Table I.22 Implied fish densities for current fish abundance and for scenarios of a 5% and a 15% risk of extinction in 100 years.

ESU		Current Abundance				Scenario 1 ^b –5% Extinction Risk				Scenario 1–15% Extinction Risk			
		Possible		Prime		Possible		Prime		Possible		Prime	
	Population ^a	Current	Historical	Current	Historical	Current	Historical	Current	Historical	Current	Historical	Current	Historical
Lower Columbia River chum	Grays	4.2	4.2			14.4	14.4			8.7	8.7		
	Lower gorge	4.6	4.6			19.7	19.5			12.3	12.2		
Upper Willamette River steelhead	North Santiam	6.6	4.0	10.7	6.1	21.4	13.0	34.8	19.8	12.4	7.5	20.1	11.4
	South Santiam	1.6	1.1	2.4	1.6	5.7	3.9	8.6	5.8	3.3	2.2	4.9	3.3
	Mollala	1.2	0.8	1.8	1.2	4.6	2.9	6.7	4.4	2.9	1.8	4.2	2.7
	Calapooia	0.5	0.3	0.7	0.5	3.4	2.2	5.0	3.2	2.0	1.3	2.9	1.8
Upper Willamette River chinook	McKenzie	6.6	4.9	22.1	16.7	20.1	14.9	67.8	51.2	11.6	8.6	39.3	29.6
	Clackamas	3.0	2.3	9.7	7.5	9.7	7.6	31.7	24.4	6.0	4.6	19.4	14.9
Lower Columbia River steelhead	Wind	5.8	5.3	7.6	6.7	26.6	24.0	34.4	30.6	16.4	14.8	21.1	18.8
	South Fork Toutle	5.7	5.0	7.7	6.8	23.3	20.6	31.5	27.9	13.5	11.9	18.3	16.2
	Sandy	3.3	2.5	4.5	3.4	11.2	8.5	15.3	11.7	6.8	5.2	9.3	7.1
	North Fork Toutle	0.8	0.5	1.2	0.8	4.3	2.7	6.3	3.9	2.9	1.8	4.2	2.6
	Kalama (winter)	4.8	4.4	6.8	6.4	19.6	18.0	27.9	25.9	11.6	10.6	16.5	15.3
	Kalama (summer)	5.7	5.3	28.2	25.0	23.2	21.7	114.6	101.6	14.1	13.3	70.0	62.1
	Hood (winter)	4.3	4.3	5.0	5.0	16.7	16.6	19.5	19.4	10.2	10.1	11.9	11.8
	Hood (summer)	5.8	5.6	22.3	21.6	21.6	21.0	83.5	81.0	13.4	13.0	51.7	50.1
	Clackamas	0.4	0.3	0.6	0.5	1.7	1.4	2.5	2.0	1.1	0.9	1.5	1.2
Lower Columbia River chinook	White Salmon	493.9	2.3		7.1	2727.3	12.7		39.0	1818.2	8.5		26.0
	Washougal	8.7	4.5	29.5	14.8	32.1	16.5	108.4	54.4	19.0	9.8	64.2	32.2
	Sandy (late fall)	5.0	4.9	16.0	15.2	16.6	16.0	52.6	49.9	10.1	9.8	32.1	30.5
	North Fork Lewis	102.9	24.5	312.1	82.1	234.2	55.7	710.8	186.8	138.5	32.9	420.2	110.4
	Mill	3.0	2.8	9.4	9.0	12.8	12.2	40.4	38.9	8.5	8.1	26.9	25.9
	Kalama	15.3	14.4	49.0	47.9	50.2	47.1	160.4	156.6	30.9	29.0	98.7	96.4
	Grays	0.5	0.5	1.4	1.4	3.8	3.8	11.0	11.0	2.3	2.3	6.6	6.6
	Elochoman	3.5	2.6	10.7	8.2	16.4	12.1	50.4	38.6	9.4	6.9	28.8	22.1
	Cowlitz	1.8	0.8	5.4	2.6	6.7	3.0	20.4	9.6	3.8	1.7	11.6	5.5
	Coweeman	6.9	6.0	21.8	19.3	29.4	25.3	92.4	81.7	18.0	15.5	56.4	50.0
	Clackamas fall	0.3	0.3	0.9	0.8	1.6	1.5	4.8	4.5	1.1	1.0	3.2	3.0
	Sandy	0.6	0.5	1.9	1.5	3.5	2.8	11.0	8.7	2.2	1.7	6.9	5.4

^a Populations and scenarios are from Appendix D (Tables D.3–D.6).^b Scenario 1 assumes no hatchery influence and no change in marine survival.

Table I.23 Implied fish densities for scenarios 2, 3, and 4 with a 5% extinction risk in the next 100 years.

ESU	Population ^a	Scenario ^b 2–5% Extinction Risk				Scenario 3 ^c –5% Extinction Risk				Scenario 4 ^d –5% Extinction Risk			
		Possible		Prime		Possible		Prime		Possible		Prime	
		Current	Historical	Current	Historical	Current	Historical	Current	Historical	Current	Historical	Current	Historical
Lower Columbia River chum	Grays	27.5	27.4			17.0	17.0			31.0	30.9		
	Lower gorge	36.9	36.6			23.4	23.2			43.0	42.7		
Upper Willamette River steelhead	North Santiam	41.4	25.1	67.2	38.2	24.7	15.0	40.2	22.8	47.1	28.5	76.5	43.5
	South Santiam	10.3	7.0	15.5	10.5	6.5	4.4	9.8	6.7	12.2	8.3	18.4	12.5
	Mollala	9.0	5.7	13.1	8.6	5.5	3.5	8.1	5.3	10.5	6.6	15.3	10.0
	Calapooia	5.4	3.5	7.9	5.0	3.4	2.2	5.0	3.2	6.4	4.1	9.4	6.0
Upper Willamette Chinook	McKenzie	37.8	28.0	127.3	96.1	23.3	17.3	78.5	59.3	45.9	34.0	154.7	116.7
	Clackamas	18.9	14.7	61.6	47.4	11.6	9.0	37.8	29.1	22.5	17.5	73.0	56.2
Lower Columbia River steelhead	Wind	49.1	44.3	63.4	56.5	30.7	27.7	39.7	35.3	59.3	53.5	76.7	68.3
	South Fork Toutle	44.1	39.1	59.8	52.9	27.0	23.9	36.5	32.3	50.3	44.5	68.0	60.2
	Sandy	21.7	16.6	29.7	22.7	12.9	9.8	17.6	13.5	25.1	19.2	34.4	26.3
	North Fork Toutle	8.1	5.1	11.9	7.4	5.3	3.3	7.7	4.8	9.1	5.8	13.3	8.3
	Kalama (winter)	35.6	32.7	50.8	47.2	20.5	18.8	29.2	27.1	41.0	37.6	58.4	54.3
	Kalama (summer)	43.7	41.0	216.4	192.0	27.0	25.4	133.7	118.6	51.5	48.3	254.6	225.9
	Hood (winter)	32.0	31.8	37.4	37.1	19.6	19.5	22.9	22.8	36.3	36.1	42.5	42.2
	Hood (summer)	43.2	42.1	166.9	162.0	25.7	25.1	99.4	96.4	49.4	48.1	190.8	185.1
	Clackamas	3.4	2.8	4.8	3.8	2.1	1.7	2.9	2.3	3.8	3.1	5.4	4.3
Lower Columbia River chinook	White Salmon	4848.5	22.6		69.4	3030.3	14.1		43.3	5454.5	25.4		78.0
	Washougal	61.9	31.7	208.8	104.7	35.7	18.3	120.4	60.4	69.0	35.4	232.8	116.8
	Sandy (late fall)	32.2	31.1	102.2	97.0	19.8	19.1	62.8	59.6	38.7	37.3	122.7	116.4
	North Fork Lewis	453.5	107.8	1376.1	361.7	274.6	65.3	833.3	219.1	552.7	131.4	1677.2	440.9
	Mill	24.7	23.6	78.1	75.1	14.5	13.8	45.8	44.1	28.1	26.9	88.8	85.5
	Kalama	97.8	91.7	312.6	305.2	56.6	53.1	181.0	176.7	111.9	105.0	357.9	349.4
	Grays	6.0	6.0	17.7	17.7	3.8	3.8	11.0	11.0	6.8	6.8	19.9	19.9
	Elochoman	29.3	21.6	90.1	69.0	17.6	13.0	54.0	41.4	32.8	24.2	100.9	77.3
	Cowlitz	12.7	5.8	38.5	18.2	7.7	3.5	23.3	11.0	14.6	6.6	44.4	21.0
	Coweeman	55.5	47.8	174.4	154.4	32.6	28.1	102.6	90.8	62.0	53.5	195.0	172.6
	Clackamas (fall)	2.8	2.6	8.6	7.9	1.8	1.6	5.4	5.0	3.2	2.9	9.7	8.9
	Sandy	6.2	4.9	19.3	15.3	4.0	3.1	12.4	9.8	7.5	5.9	23.4	18.5

^a Populations and scenarios are from Appendix D (Tables D.3–D.6).

^b Scenario 2 assumes some hatchery influence and no change in marine survival.

^c Scenario 3 assumes no hatchery influence and a change in marine survival.

^d Scenario 4 assumes both a hatchery influence and a change in marine survival.

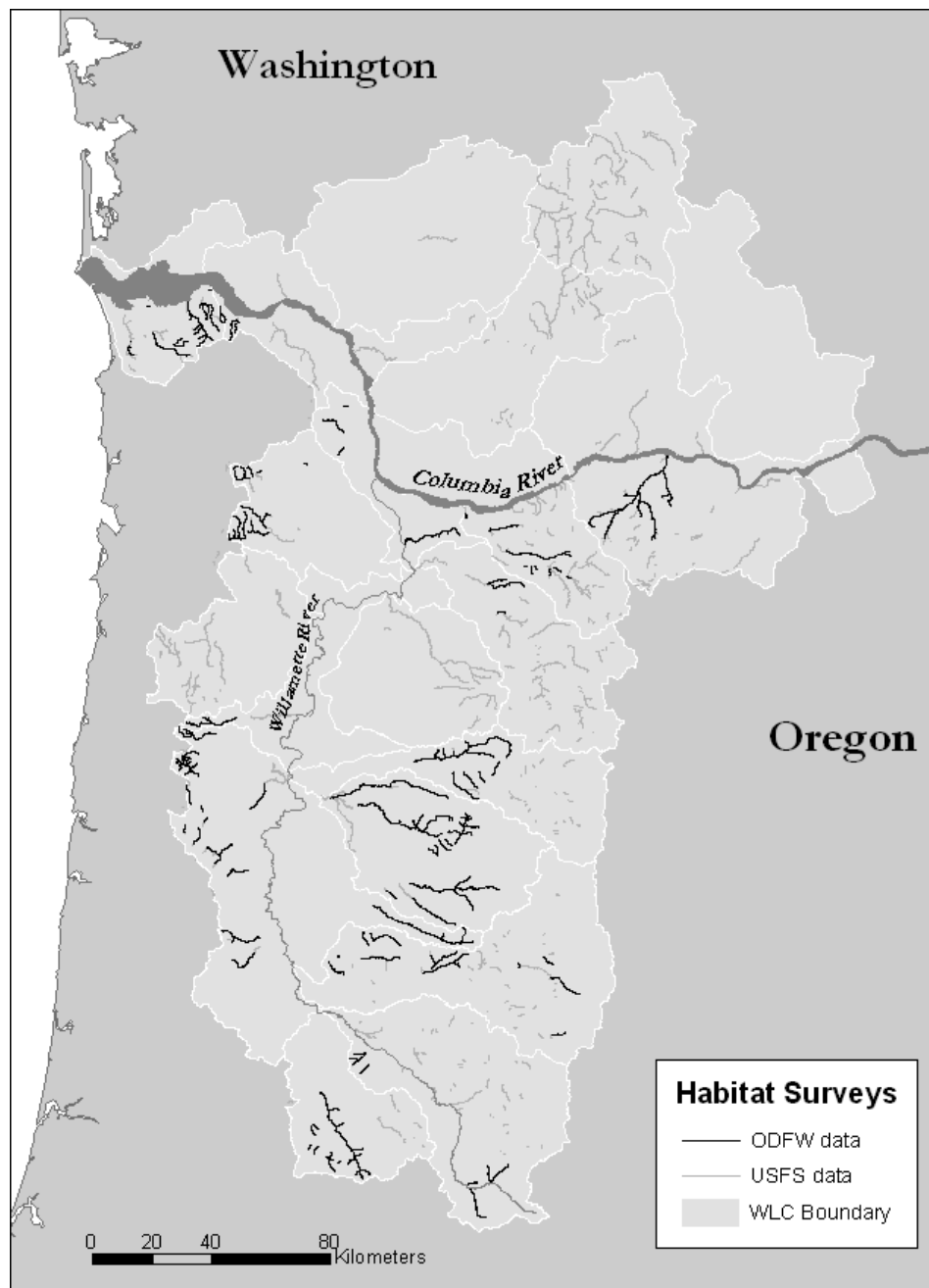


Figure I.1 Reaches in the Willamette/Lower Columbia domain with existing habitat digital survey data from the Oregon Department of Fish and Wildlife (ODFW) and the U.S. Forest Service (USFS). The ODFW digital survey data was used for the channel-width modeling. Please note that neither survey includes the mainstem Willamette or Columbia Rivers.

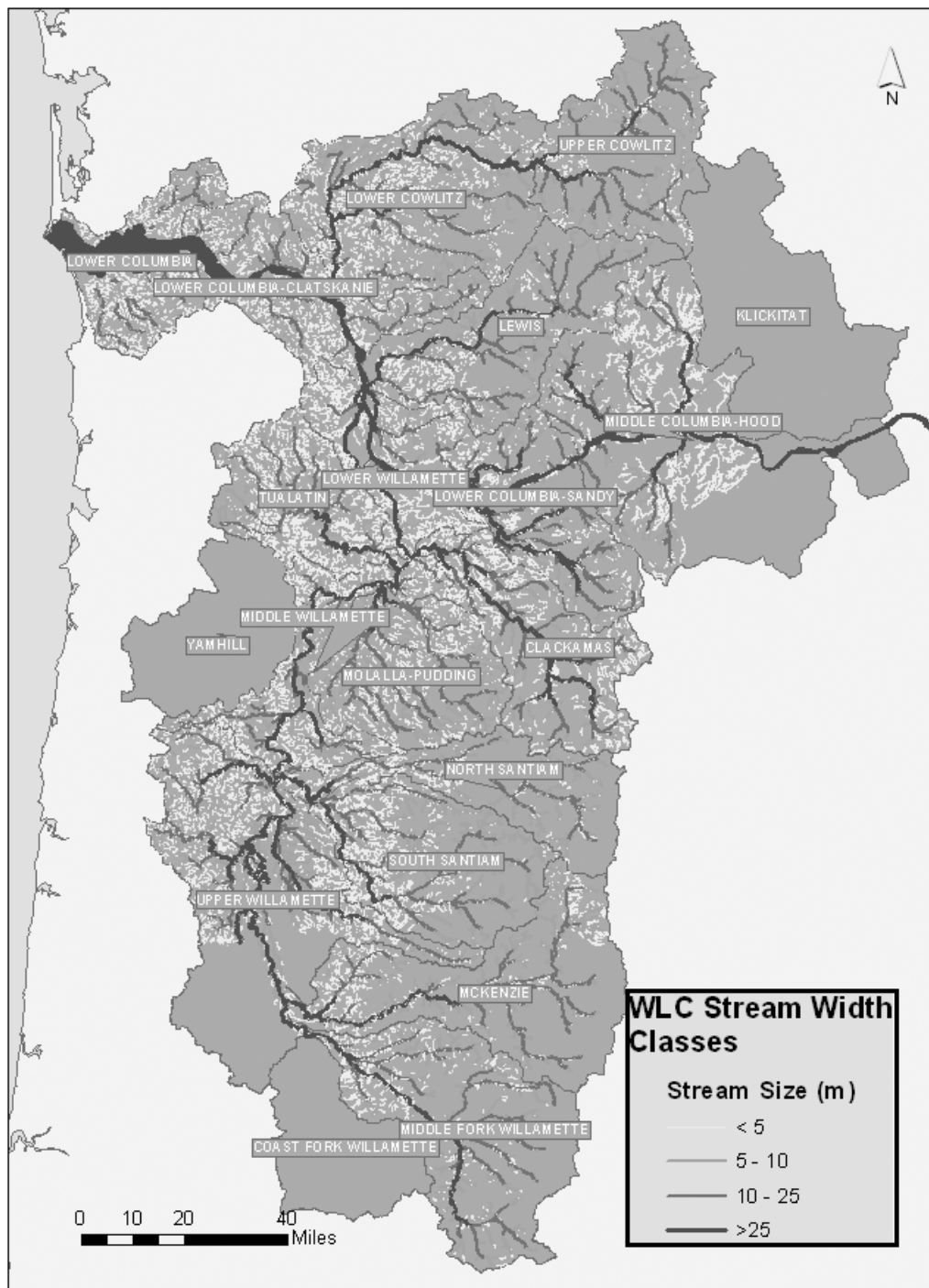


Figure I.2 Modeled widths for the Willamette/Lower Columbia domain, divided into four size classes.

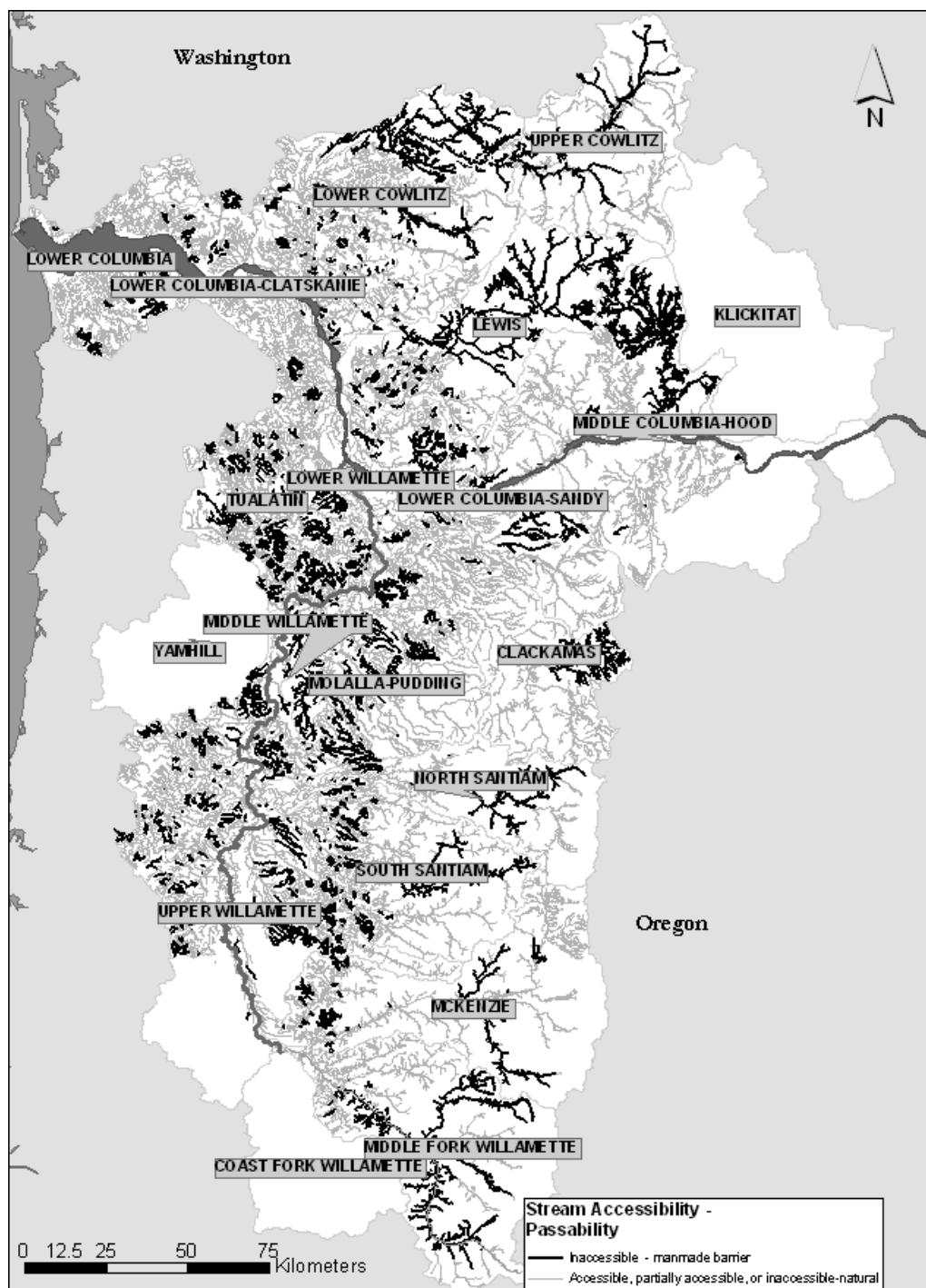


Figure I.3 Stream accessibility and passability for all streams considered in the Willamette/Lower Columbia analysis. Legend describes the various categories of accessibility. Stream kilometers that are inaccessible because of man-made barriers are indicated in black. Light outlines and labels indicate the fourth-field hydrologic basin.

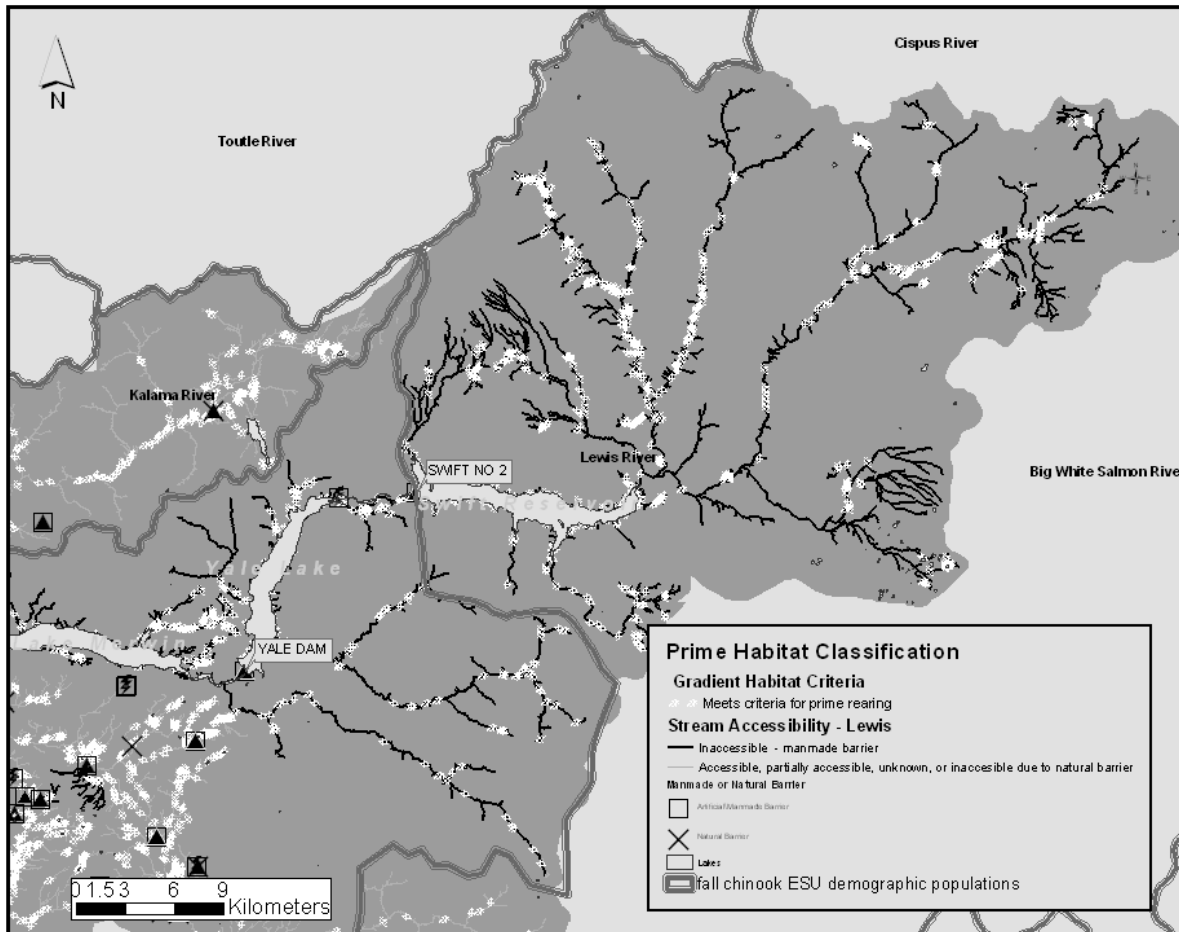


Figure I.4 Example of the identification of prime and possible habitat attributes. Map indicates stream reaches classified as “prime habitat” for chinook rearing or spawning in the Lewis River, based only on defined gradient thresholds. The white symbols indicate patches of streams (reaches) that meet the thresholds. Black streams represent streams inaccessible due to man-made barriers.

Willamette/Lower Columbia Salmonid Viability Criteria

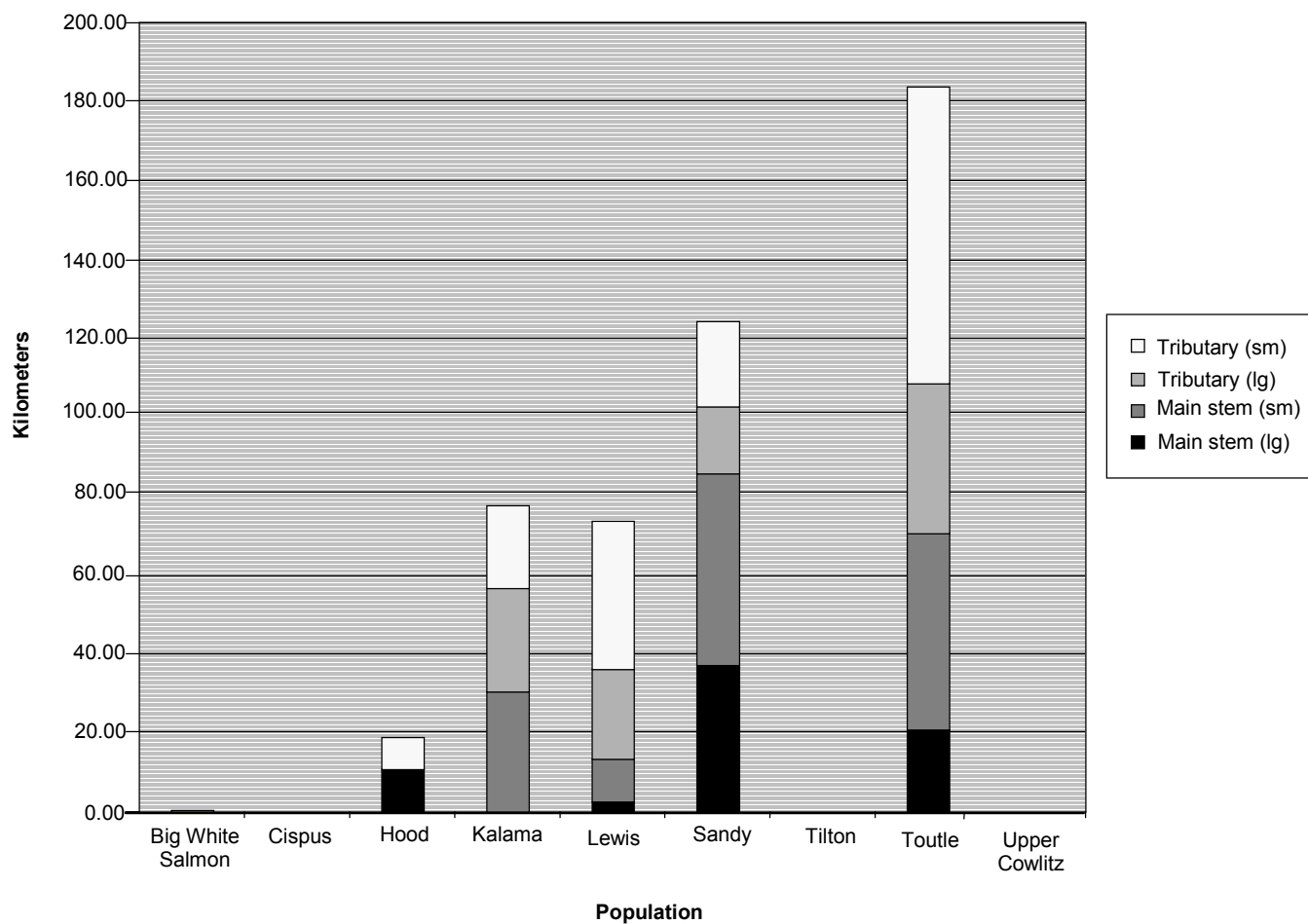


Figure I.5 Currently available kilometers of possible spawning habitat for Lower Columbia spring chinook salmon populations.

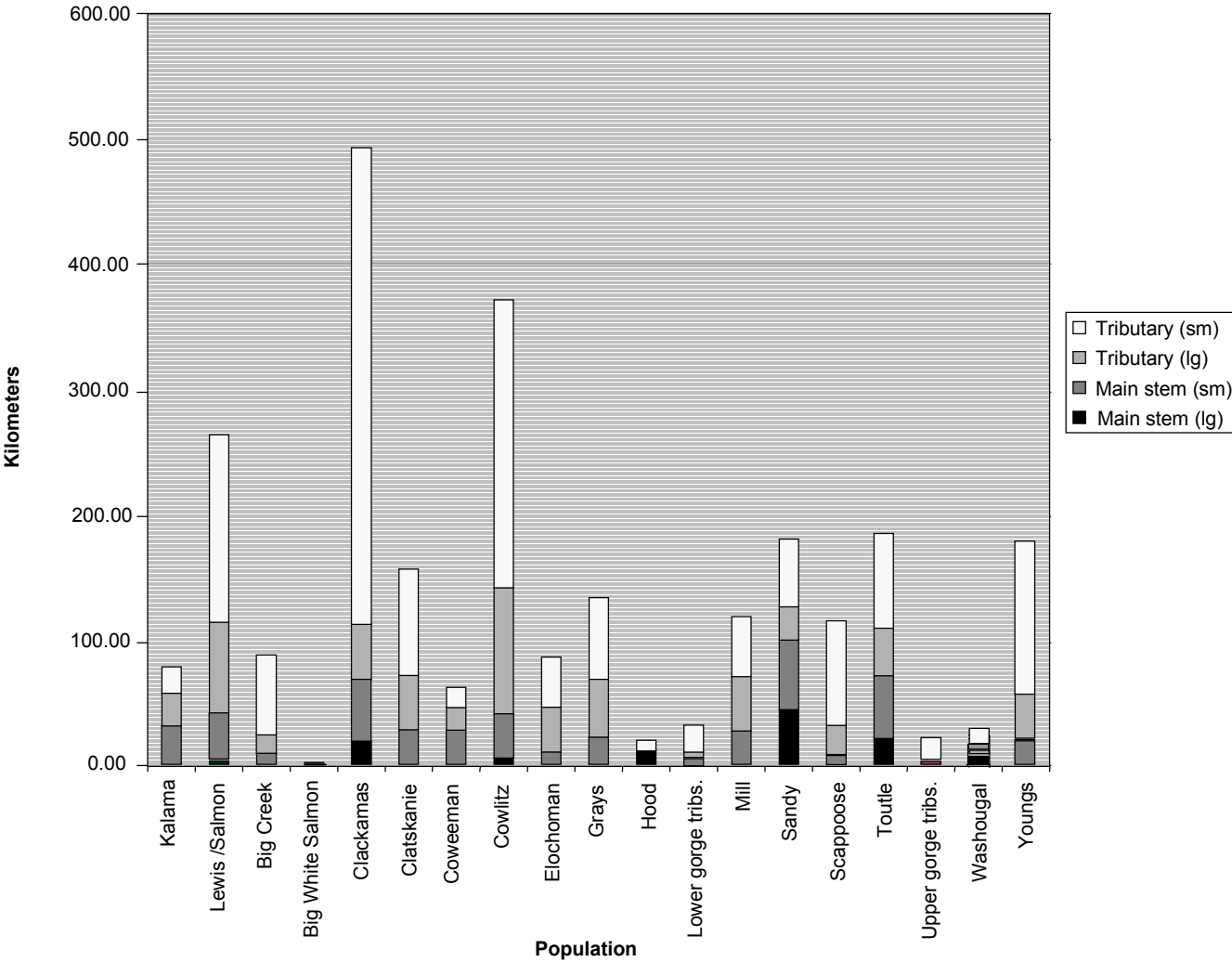


Figure I.6 Currently available kilometers of possible spawning habitat for Lower Columbia fall chinook salmon populations.

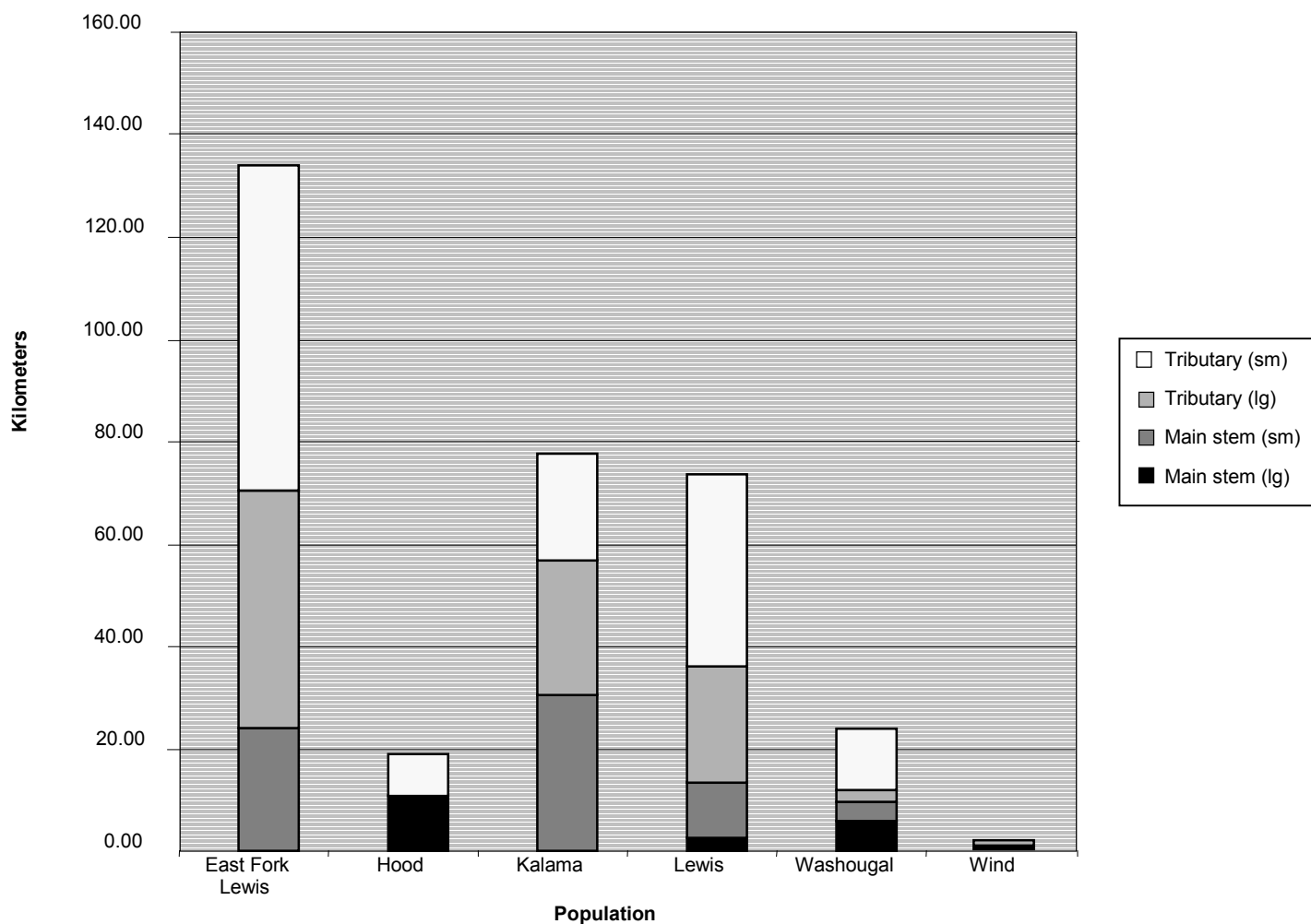


Figure I.7 Currently available kilometers of possible spawning habitat for Lower Columbia summer steelhead salmon populations.

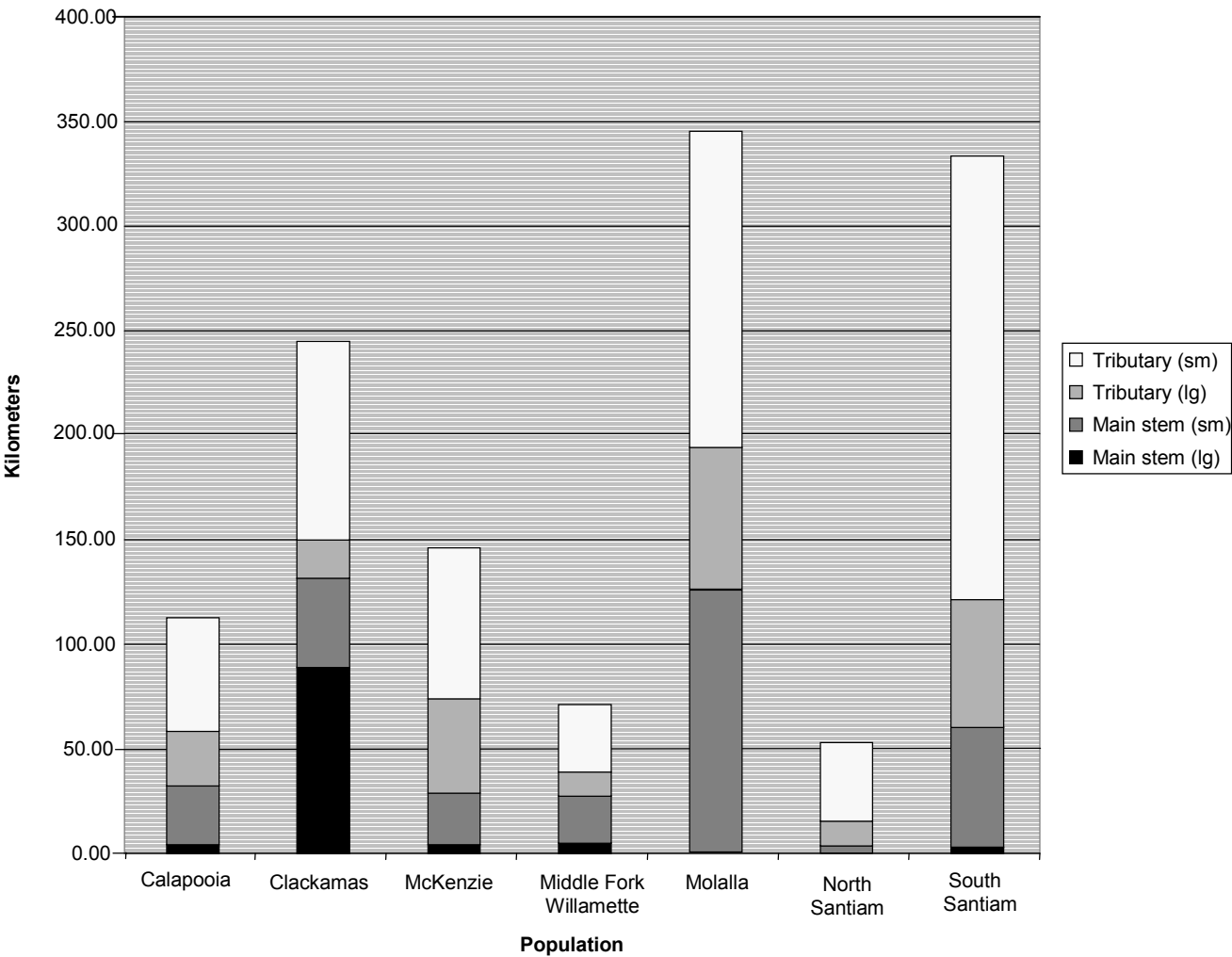


Figure I.8 Currently available kilometers of possible spawning habitat for Willamette spring chinook salmon populations.

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APPENDIX J

HPVA RESULTS FOR SALMON AND STEELHEAD PRODUCTION IN WASHINGTON LOWER COLUMBIA BASINS

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Tables J.1 through J.5 present the results of habitat population viability analyses (HPVA) completed to date of Lower Columbia basins in Washington State for chinook salmon, steelhead, and chum salmon production. Results are presented for three levels of habitat attributes: current conditions, PFC+ conditions, and historical conditions. The PFC+ level assumes properly functioning conditions in freshwater, as per the Matrix of Pathways and Indicators (NMFS 1996), and historical estuarine and nearshore conditions. Productivities, capacities, and equilibrium (no harvest) numbers of spawners (Neq) are presented for each of these levels of habitat condition, assuming a Beverton-Holt production function. The diversity index is the percentage of theoretical life-history trajectories possible that are viable under the specified habitat conditions.

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Table J.1 HPVA results for fall chinook salmon production in Washington Lower Columbia basins.

Basin	Habitat Condition	Productivity	Capacity	N_{eq}	Diversity Index
Grays	Current	2.4	854	491	92%
	PFC+	10.2	2,485	2,241	100%
	Historical	12.6	2,690	2,477	100%
Lower Cowlitz	Current	5.9	9,626	7,992	65%
	PFC+	14.6	39,931	37,191	100%
	Historical	16.2	57,506	53,956	100%
Coweeman	Current	3.6	2,355	1,695	100%
	PFC+	12.1	4,885	4,482	100%
	Historical	16.0	5,304	4,971	100%
Toutle	Current	2.1	6,974	3,715	56%
	PFC+	11.1	23,840	21,683	100%
	Historical	14.5	27,275	25,392	100%
Kalama	Current	2.3	1,007	560	73%
	PFC+	5.9	1,519	1,262	100%
	Historical	9.8	2,733	2,455	100%
East Fork Lewis	Current	4.0	1,842	1,375	100%
	PFC+	10.9	3,980	3,614	100%
	Historical	13.7	4,554	4,220	100%
North Fork Lewis	Current	11.5	14,764	13,483	100%
	PFC+	19.2	36,226	34,339	100%
	Historical	22.7	45,369	43,371	100%
Washougal	Current	4.7	2,592	2,041	100%
	PFC+	12.4	6,260	5,756	100%
	Historical	16.9	7,990	7,518	100%
Wind	Current	4.1	647	490	65%
	PFC+	9.5	2,642	2,363	97%

Table J.2 HPVA results for spring chinook salmon production in Washington Lower Columbia basins.

Basin	Habitat Condition	Productivity	Capacity	N_{eq}	Diversity Index
Toutle	Current	1.2	465	68	2%
	PFC+	15.3	2,280	2,131	100%
	Historical	25.1	3,022	2,901	100%
Kalama	Current	3.6	1,186	852	86%
	PFC+	12.5	2,409	2,217	100%
	Historical	26.9	4,340	4,178	100%
Wind	Current	3.6	1,747	1,256	89%
	PFC+	7.8	2,567	2,238	100%
	Historical	17.7	4,215	3,977	100%

Table J.3 HPVA results for summer steelhead production in Washington Lower Columbia basins.

Basin	Habitat Condition	Productivity	Capacity	N_{eq}	Diversity Index
Kalama	Current	3.9	1,237	923	88%
	PFC+	9.5	1,606	1,437	97%
	Historical	16.8	3,365	3,165	100%
East Fork Lewis	Current	1.8	200	91	17%
	PFC+	4.4	395	306	85%
	Historical	7.2	491	422	98%
Washougal	Current	3.0	568	376	67%
	PFC+	8.9	1,227	1,089	91%
	Historical	15.4	1,517	1,419	100%
Wind	Current	5.9	1,357	1,125	71%
	PFC+	8.7	1,588	1,404	86%
	Historical	14.0	2,463	2,288	100%

Table J.4 HPVA results for winter steelhead production in Washington Lower Columbia basins.

Basin	Habitat Condition	Productivity	Capacity	N_{eq}	Diversity Index
Lower Cowlitz	Current	5.0	397	318	30%
	PFC+	18.6	1,639	1,551	99%
	Historical	28.4	1,733	1,672	94%
Coweeman	Current	3.6	1,094	786	50%
	PFC+	12.6	1,894	1,743	92%
	Historical	18.4	2,372	2,243	100%
Toutle (total)	Current	2.8	2,145	1,382	37%
	PFC+	15.8	8,003	7,497	96%
	Historical	23.7	9,116	8,731	98%
South Fork Toutle	Current	3.4	785	551	52%
	PFC+	15.6	2,344	2,194	91%
	Historical	21.9	2,752	2,627	99%
Main stem/North Fork Toutle	Current	1.5	274	96	7%
	PFC+	15.0	2,119	1,978	100%
	Historical	24.9	2,370	2,275	100%
Green	Current	2.3	525	291	33%
	PFC+	10.2	1,344	1,212	96%
	Historical	17.1	1,588	1,495	100%
Kalama	Current	5.2	1,314	1,060	92%
	PFC+	16.1	1,793	1,681	99%
	Historical	22.6	579	554	100%
East Fork Lewis	Current	5.3	1,337	1,087	61%
	PFC+	17.3	2,633	2,480	98%
	Historical	26.6	3,253	3,131	100%
North Fork Lewis	Current	5.0	416	333	96%
	PFC+	19.8	649	616	100%
	Historical	30.2	737	713	100%
Washougal	Current	3.9	697	519	52%
	PFC+	18.2	2,111	1,995	100%
	Historical	28.7	2,588	2,497	100%
Duncan	Current	5.3	69	56	74%
	PFC+	10.1	132	119	93%
	Historical	27.6	242	233	93%
Hardy	Current	4.4	13	10	55%
	PFC+	6.1	19	16	59%
	Historical	11.3	31	28	99%
Hamilton	Current	5.7	265	219	39%
	PFC+	14.0	312	290	100%
	Historical	32.6	549	532	100%
Wind	Current	5.1	112	90	80%
	PFC+	12.4	192	177	100%
	Historical	20.2	256	243	100%

Table J.5 HPVA results for chum salmon production in Washington Lower Columbia basins.

Basin	Habitat Condition	Productivity	Capacity	N_{eq}	Diversity Index
Grays	Current	3.5	1,512	1,080	2%
	PFC+	6.4	7,859	6,639	67%
	Historical	9.2	8,431	7,511	72%
Lower Cowlitz	Current	1.5	5,861	1,819	6%
	PFC+	4.5	110,837	86,424	100%
	Historical	5.9	158,702	131,803	100%
Coweeman	Current	0.0	334	0	0%
	PFC+	4.2	7,771	5,921	54%
	Historical	4.5	10,406	8,072	58%
Toutle	Current	0.0	564	0	0%
	PFC+	4.8	15,937	12,630	56%
	Historical	7.8	22,613	19,707	61%
Kalama	Current	1.1	1,462	157	2%
	PFC+	4.7	7,879	6,210	43%
	Historical	6.6	11,743	9,953	50%
East Fork Lewis	Current	0.0	4,215	0	0%
	PFC+	5.1	30,269	24,334	30%
	Historical	7.6	41,924	36,415	40%
North Fork Lewis	Current	1.7	5,375	2,157	20%
	PFC+	4.2	41,051	31,183	67%
	Historical	6.6	62,851	53,256	72%
Washougal	Current	1.0	568	17	0%
	PFC+	6.2	9,939	8,336	100%
	Historical	8.6	17,129	15,140	100%
Duncan	Current	3.7	111	81	100%
	PFC+	9.5	241	215	100%
	Historical	10.3	47	43	100%
Hardy	Current	3.0	94	63	36%
	PFC+	10.4	245	221	36%
	Historical	10.7	277	251	42%
Hamilton	Current	2.9	852	557	78%
	PFC+	10.0	3,031	2,729	88%
	Historical	10.7	3,140	2,847	89%
Wind	Current	1.6	952	349	13%
	PFC+	5.2	6,432	5,203	57%
	Historical	6.0	10,713	8,912	64%

APPENDIX K

CATASTROPHIC RISK ASSESSMENT OF LOWER COLUMBIA AND WILLAMETTE RIVER ESUs FOR ENDANGERED AND THREATENED PACIFIC SALMON

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Introduction

Catastrophic events are among the factors, along with long-term demographic processes and evolutionary potential, that need to be considered when relating viable salmonid populations (VSPs) to viable evolutionarily significant units (ESUs). Catastrophes are sudden (within-season) events that eliminate or severely reduce (by up to 75%) adult populations (McElhany et al. 2000). A variety of natural and anthropogenic catastrophic events occurring intermittently over evolutionary time scales can have long-term consequences. If preceded by gradual climatic change or overfishing, these events may result in ecosystem shifts (Scheffer et al. 2001). Natural catastrophes include volcanoes, earthquakes, floods, landslides, extreme weather (droughts), unusual fires, and disease epidemics. Anthropogenic catastrophic events include oil/chemical spills, dam construction or diversion/dam failure, floods, disease epidemics from hatcheries, and major miscalculations in harvest. Some catastrophic events can also result from the interaction of natural and anthropogenic factors.

The number of threats facing salmonid populations suggests that catastrophes may have a substantial influence on extinction risk. Salmon have enhanced their long-term stability in the face of ice ages, continental uplifts, and volcanic eruptions by maintaining diverse populations, habitats, and life-history diversity, thus spreading risk and providing redundancy (Levin and Schiwe 2001). The risk of extinction posed by catastrophic events for an entire ESU can be estimated by evaluating risk for separate populations (McElhany et al. 2000) as well as for nearby populations (correlated risks).

Catastrophic events are not commonly considered a part of species listing or recovery plans. Of 181 recovery plans reviewed by a National Center for Ecological Analysis and Synthesis (NCEAS) working group, 13 ($\approx 7\%$) cited catastrophes/stochastic events as a factor in the listing decision, and 57 ($\approx 31\%$) cited catastrophes/stochastic events as a factor in the recovery plan. (www.nceas.ucsb.edu/recovery/data). Only 31 ($\approx 17\%$) listed catastrophes/stochastic events as a major threat; however, 51% of those plans assigned the highest implementation priority to tasks that address these factors. Catastrophic events are of primary importance in a small number of cases; for example, the recovery plan for the federally listed sea otter in California identified catastrophic oil spills as the primary risk to population viability, with quantitative estimates of risks from oil spills forming the basis of the recovery goals (Ralls et al. 1996).

While catastrophic events vary in frequency, scope, and impact, they share features that make them amenable to quantification and of potential importance for salmon populations. This document investigates a variety of natural and anthropogenic catastrophes in order to make quantitative and qualitative assessments of catastrophic risk for threatened and endangered Pacific salmonid ESUs in the Lower Columbia and Upper Willamette Rivers, specifically chinook salmon (*Oncorhynchus tshawytscha*), steelhead trout (*O. mykiss*), and chum salmon (*O. keta*). Herein, we analyze catastrophic risks from volcanoes, glacial outbursts, earthquakes, landslides, disease epidemics from hatchery operations, and transportation oil/chemical spills. Risks from floods, fire, pollution from oil/chemical storage, and from land use (industrial zoning, pesticide use) are being analyzed to more fully understand the suite of catastrophic risks that exist for these endangered Pacific salmonid ESUs in the Lower Columbia and Upper Willamette Rivers.

Volcanoes and Glacial Outbursts

Volcanoes

Volcanoes and flows of water, mud, and debris associated with glaciers pose a considerable risk to populations in watersheds that emanate from the chain of volcanic mountains in the Cascade Mountains. In fact, Mount Rainier is considered an extremely dangerous volcano (Perkins 2001). Volcanic activity epitomizes extreme unpredictability—catastrophic events that may be statistically predictable, but only in time intervals much longer than the generation time of salmonids (Thorpe 1994). These catastrophic risks have an occurrence interval of 100–1,000 years (Bisson et al. 1997), and they can have devastating consequences for salmonids, especially in watersheds close to an eruption. Volcanism can result a variety of chemical and physical alterations, including increased delivery of fine sediments and organic matter, scouring of channels from mudflows, formation of mudflow terraces along rivers, destruction of riparian vegetation, damming of streams, and the potential creation of new lakes (NRC 1996). The effects on the salmon's habitat include sedimentation of spawning gravels, loss of pool habitats from mudflows, short-term lethal levels of sediment and temperature during eruptions, and formation of migration blockages. Potential positive effects include creation of pool habitat in areas with tree blowdowns, creation of new overwintering habitat and side channels along mudflow terraces, and long-term benefits to lake-dwelling species (NRC 1996). Physical, biological, and chemical changes resulting from even modest volcanic eruptions can be extreme (Dorava and Milner 1999).

The 18 May 1980 eruption of Mount St. Helens in 1980 provides examples of the potential short and long-term consequences of volcanic activity. The effects of the Mount St. Helens eruption were dramatic and variable. The eruption damaged over 500 km² of forest and riparian vegetation, sending water temperatures soaring to 26°C on some streams (Lucas 1986), and increasing hillside erosion, due to a lack of groundcover, contributed to stream bedloads. The resultant debris flow extended 24 km down the North Fork Toutle River, and mudflows buried stream channels in the North and South Fork Toutle Rivers, eliminating fish habitat (Lucas 1986). With the North Fork Toutle River buried to an average depth of 47 m (maximum 183 m), and smaller mudflows in the South Fork Toutle River and parts of the upper Lewis River and Kalama River, increased water levels flooded the Toutle and Cowlitz River Basins. Mudflow

deposits that clogged the channels of the Cowlitz River also led to water temperatures in the lower reaches exceeding 32°C. Lahars reduced the flood-stage capacity at Castle Rock on the Cowlitz River from 76,000 cubic feet per second (cfs) to less than 15,000 cfs, and reduced channel depth on the Columbia River (12 to 4.25 m), stranding 31 ships in upstream ports (Leider 1989).

Effects of the Mount St. Helens eruptions on salmonids were dramatic and variable; recovery of salmonid populations after volcanic eruptions can potentially be relatively quick due to temporary food abundances, recolonization potential, and relative scarcity of predators and competitors (Bisson et al. 1997). However, the majority of aquatic life in the Toutle River watershed was probably eliminated immediately (Leider 1989). Additional fatalities included approximately 12 million salmon fingerlings in hatcheries (Brantley and Myers 2000) as well as increased summer and winter mortality of stocked juvenile coho salmon related to high stream temperatures and the lack of large organic debris, respectively (Martin et al. 1986). Production of stocked juvenile coho in three impacted third-order streams increased annually, peaked six years after the eruption, and declined to normal thereafter (Bisson et al. 1988, 1997). Reduced invertebrate communities resulted in food impacts for subyearling chinook in the Columbia River estuary in 1980 and 1981, although effects on food resources were believed to be short-term (Kirn et al. 1986). Increased straying of fish to unaffected streams and rivers can result from blocked access to spawning grounds; many fish initially avoided the Toutle River and strayed extensively into other Columbia River tributaries (Lucas 1986), and the percentage of nonnatal steelhead in unaffected Columbia River tributaries increased from 16% to 45% (both winter and summer runs) (Leider 1989). Straying to the Kalama River and the North Fork Lewis River was also extensive, and the decline in fish numbers in the Cowlitz River continued to 1983–1984 before reverting to pre-eruption levels (Leider 1989).

Recent eruptions in a chain of volcanoes west of Cook Inlet, Alaska, further exemplify the direct and indirect effects on salmonids, including changes to water quality, channel geometry, and riparian vegetation (Dorava and Milner 1999). During the 1989–1990 Redoubt Volcano eruption, the riparian zone was removed or killed in place by lahars, which reduced allochthonous input and, subsequently, primary and secondary production. Such habitat decreases or degradation can persist for years, and subsequent effects include initial migratory impediments from lahar deposits, unstable streambeds, and silt in spawning gravel beds (Dorava and Milner 1999). Salmon populations were likely seriously affected due to changes to fish access. Thick deposits of fine sediment modified large channels and spawning sites, and food sources were eliminated for rearing fish. Being washed out by mud and debris flows likely immediately killed juveniles and affected future spawning activities. Food sources were subsequently eliminated for rearing fish. Although macroinvertebrate communities can recover as early as five years after a major volcanic eruption, it is not clear whether these are stable communities. Upstream sources of macroinvertebrates can hasten a return to normalcy as well, but comparison among volcanoes is problematic—macroinvertebrate recolonization at Mount St. Helens began rapidly but took place over a long time.

Glacial Outbursts

Glacial outbursts, from the sudden release of water stored within or at the base of glaciers, pose a serious hazard in river valleys on volcanoes. Glacial outbursts at Mount Rainier can be unrelated to volcanic activity, and the peak discharge of outbursts may be greater than that of extreme meteorological floods. At least 36 outburst floods have been recorded from the Kautz, Nisqually, South Tahoma, and Winthrop Glaciers on Mount Rainier during the twentieth century, destroying bridges, roads, and Mt. Rainier National Park visitor facilities 10 times since 1926. Well-studied outbursts—from South Tahoma Glacier—are correlated with periods of unusually high temperatures or heavy rain in summer or early autumn (Hoblitt 1998). Many of these glacial outburst floods transform to lahars by incorporating large quantities of sediment from channel walls and beds; consequently, they are included with lahars for purposes of hazard zonation (Hoblitt 1998). Discharges of water and debris have also occurred at Mount Hood, resulting in significant losses of salmonid spawning and rearing habitat on the East Fork of the Hood River (Kostow et al. 2000).

Earthquakes

Earthquakes, and secondary landslide hazards associated with ground motion, pose a risk to populations in watersheds associated with the offshore Cascadia subduction zone and to some extent with the Cascade Range volcanoes. Little literature exists on direct or indirect effects of earthquakes on Pacific salmonids. The hazard posed by ground shaking and related secondary damage to watersheds, and the fish habitat contained therein, has been estimated and can be used as a proxy for damage to fish resources.

Each year more than 1,000 earthquakes are recorded in Washington State, with 15 to 20 causing substantial ground shaking. Destructive earthquakes occur much less often; the last earthquake to cause widespread damage in Washington occurred in 1965. Larger earthquakes may have occurred every several hundred or thousand years in the Pacific Northwest; the most recent such earthquake occurred about 300 years ago (Noson et al. 1988). The effects of earthquakes include burial of nearby valley floors; such an avalanche was triggered by a moderate (magnitude 5) earthquake, which followed eight weeks of intense seismic activity beneath Mount St. Helens during the 18 May 1980 volcanic eruption (Noson et al. 1988).

Earthquakes may result in secondary damage from landslides (see “Landslides” below), such as the 14 earthquakes large enough to trigger landslides in Washington from 1872 to 1980. Earthquakes on Mount Rainier, Mount St. Helens, and around Puget Sound have been known to trigger landslides, and ground shaking produced by earthquakes can weaken and collapse bluffs. Future earthquakes in Washington are expected to generate more landslides and greater habitat changes than those reported for past earthquakes (Noson et al. 1988).

Earthquakes off the Pacific Northwest coast may also result in tsunamis when large, rapid movements in the seafloor displace the water column above, thus setting off the destructive waves. Offshore tsunamis can strike adjacent shorelines within minutes and cross the ocean at speeds of up to 600 mph. A 1946 earthquake in the Aleutian Islands of Alaska initiated a tsunami that reached Hawaii in less than five hours, had waves as high as 55 feet, and killed 173 people (Manson and Walkling 1998). A dozen very large earthquakes (magnitudes ≥ 8) have occurred in

the Cascadia subduction zone. On the Pacific Northwest coast, risks exist from distant and local tsunamis and computer models indicate that tsunami waves generated by local events might reach 55 feet and affect the entire coastal region (Manson and Walkling 1998).

Landslides

Washington has many sites susceptible to landslides, including steep rocky slopes along the Columbia Gorge and rugged terrain in the Cascade Mountains. Although landslides are propelled by gravity, they can be triggered by geologic or anthropogenic forces. Volcanic eruptions can initiate earth movement on a grand scale, particularly lahars, mixtures of volcanic ash and water. Cascade volcanoes offer many sites for rock and ice avalanches, rock falls, and debris flows on their steep slopes. They are particularly vulnerable to landslides because of the layered and jointed volcanic rocks lying parallel to the mountain slopes, weakened by the effects of steam and hot groundwater and oversteepened by erosion. In addition, icefalls from glaciers can trigger landslides, and snow and ice add to the mobility of such slides.

The 1949 Olympia earthquake generated more than 20 landslides, as far as 180 km from the epicenter; the 1965 Seattle/Tacoma earthquake generated 21 landslides, as far as 100 km from the epicenter. Fourteen earthquakes from 1872 to 1980 are known to have triggered landslides in Washington. Landslides on Mount Rainier were reported for earthquakes in 1894, 1903, and 1917, and a massive 2.8-km³ rockslide/debris avalanche on the north side of Mount St. Helens during the catastrophic eruption of 18 May 1980 was triggered by a moderate (magnitude 5) earthquake that followed eight weeks of intense earthquake activity beneath the volcano. Sudden water displacement from landslides can also generate destructive water waves, such as occurred when a 300-foot bluff along the Tacoma Narrows, weakened by the 1949 earthquake, collapsed into Puget Sound three days later. Future earthquakes in Washington are expected to generate more landslides than were reported for the 1949 or 1965 earthquakes, when rainy-season precipitation was near or below average throughout the Puget Sound area.

Earthquakes notwithstanding, the major causes of landslides in the Northwest are continuous rains that saturate soils. Mud and debris flows are frequently the direct consequence of human activity. Seemingly insignificant modifications of surface flow and drainage may induce landslides, and building placement may lead to the loss of structures. In Portland, population pressure has pushed housing and highway construction into landslide-prone areas, where improper drainage induces disastrous sliding. Landslides result from agricultural irrigation and clearcutting of forests from naturally steep slopes. A 1996 Forest Service study of 244 landslides found that 91 were associated with logged-over lands, 93 with roads, and 59 in undisturbed forests; the combination of logging and road-building increases slide frequency fivefold over a 20-year period compared to undisturbed forested lands. Most of the 250 landslides in the Clackamas River watershed and in the Mount Hood National Forest during the floods of 1996 were in lands logged over or criss-crossed by dirt logging roads (<http://www.oregongeology.com/landslide/landslidehome.htm>).

The impact of landslides on stream drainages and reservoirs can pose significant risk to downstream areas. Landslides or debris flows into reservoirs or lakes may displace enough water to cause severe downstream flooding, and water ponded behind landslide-debris dams can cause severe floods when these natural dams are suddenly breached. Such outburst floods are most

likely near volcanic centers active within the past 2 million years; the Toutle River was blocked by a debris flow triggered by an earthquake during the 1980 eruption of Mount St. Helens. The debris flow dam raised the level of Spirit Lake by 60 m, requiring the U.S. Army Corps of Engineers to construct a tunnel through bedrock to lower the lake level and reduce flood danger from a sudden release of water (Crandell and Mullineaux 1978).

Disease Epidemics from Hatchery Operations

Characteristics of host-pathogen interactions make disease epidemics potentially catastrophic events. Host-pathogen interactions are understudied and often indiscernible in the wild, and disease epidemics occur seemingly out of nowhere. While documentation of chronic background levels of diseases provides information on the past, future risks from new and deadly pathogens/parasites may be unrelated to past events. Moreover, while there is much information on disease incidence and effects on hatchery salmon and steelhead, our understanding of the effects of disease on released hatchery fish and interaction with wild fish is not well understood (NRC 1996).

Fish and pathogen strains have generally co-evolved in a way that common microorganisms do not kill the hosts (Schreck 1996). However, mutations may result in abnormally virulent strains and a series of mutual population expansions and crashes, not unlike predator/prey interactions. Epidemics may burn themselves out because hosts die, and thus pathogen density decreases and cannot be transmitted effectively, or there may be no loss of virulence, only a low rate of infection. Although it is recognized that pathogens reside in wild populations and that diseases are transferred between wild and hatchery fish, the initial introduction of pathogens into a population results largely from infected fish being moved by humans into susceptible populations (Reno 1998).

Variable susceptibility to pathogens is, in part, inherited, and wide variation exists in response to pathogen challenge in wild fish because of their wide genetic background (Anderson 1996). Individual differences have been identified in wild stocks and among hatchery strains (Beacham and Evelyn 1992). The variable susceptibility among fish might be considered normative in contrast to genetically selected hatchery strains, which might have lost much of this variability. While genetically diverse and spatially separated wild fish may be able to deny new or old virulent strains the opportunity to proliferate through close contact, hatchery fish, being possibly less diverse and more densely packed, may transmit a virulent strain that would otherwise subside as a result of competition with less-virulent counterparts (Coutant 1998).

Fish culture may strongly influence the number of asymptomatic carriers compared with their numbers in wild conditions (Coutant 1998). These carriers are common (Anderson 1990), and they can transmit pathogens to susceptible fish that they encounter. In wild fish, culling from disease early in life may be masked, because we expect fairly high early-life-stage mortality. However, prophylactics in hatchery culture situations may reduce progressive early mortality. A high loss of hatchery outplants, once dispersed in the wild, may merely be an expression of the delayed culling process through disease. The long-term effects for surviving wild and cultured fish might be equivalent, but the development of the disease(s) from the perspective of the pathogen's normal ecology might be very different between infection of young fish and of older ones held in a hatchery for a year before release. The pathogen might thereby be held to its

normative cycle of attack, infection, growth, and dispersal in early juveniles rather than creating an abnormal disease cycle with older fish (Coutant 1998).

In hatchery salmonids, the negative effect of rearing density on growth, condition factor, food conversion efficiency, as well as increased physiological stress and rates of mortality, have been extensively documented (Flagg et al. 2000). What role disease plays in these reduced performance measures remains unclear, however reductions in performance measures due to diseases, which themselves are related to rearing densities, are well documented. For example, infection by bacterial kidney disease (BKD), a major pathogen in Pacific Northwest fish hatcheries, significantly reduced the ability of juvenile chinook to avoid fish predators in laboratory experiments (Mesa 1998). The potential effect of high densities of salmon in hatchery facilities throughout the Pacific Northwest can be seen in surveys of hatchery disease (Table K.1). Pathogen detection varied from relatively rare at all facilities, such as with viral hemorrhagic septicemia (VHS), to extremely common, such as with BKD.

Despite improvements in hatchery disease management, many of these diseases continue to be chronic problems for salmonids in hatchery facilities. Disease mortality rates in hatchery populations can be very high, depending on environmental conditions, and vary considerably among pathogens (Table K.2).

Table K.1 Facilities (% in state, agency, or tribal hatcheries) testing positive for the major salmonid diseases (1998-1993).^a

	Viral				Bacterial				Parasitic			
	IHN	IPN	VHS	EIBS	BKD	FUR	ERM	CWD	PKD	MC	CS	ICH
Oregon state	18.1	0.3	0.0	24.6	53.1	35.9	17.8	84.8	0.0	2.9	33.3	26.2
Washington	11.5	0.7	0.1	34.2	52.6	20.1	17.0	60.3	3.5	0.0	11.9	24.4
U.S. Fish and Wildlife Service	37.5	1.0	0.0	27.2	84.9	23.7	20.0	34.9	0.0	0.6	30.6	24.0
Northwest Indian Fisheries Commission	2.9	0.0	0.6	ns ^c	51.5	14.0	18.1	39.9	56.3	0.0	0.0	15.0
Average ^b	20.2	1.3	0.2	14.5	50.3	17.8	15.0	36.8	12.5	4.4	18.8	20.8

^a Adapted from Waknitz (2002), based on PNWFHPC 1993 report.

^b Average values, including data from Alaska, Canada, Idaho, and Montana.

^c NS = not surveyed

Key:

IHN	Infectious hematopoietic necrosis
IPN	Infectious pancreatic necrosis
VHS	Viral hemorrhagic septicemia
EIBS	Erythrocytic inclusion body syndrome
BKD	Bacterial kidney disease
FUR	Furunculosis
ERM	Enteric redmouth disease
CWD	Coldwater disease
PKD	Proliferative kidney disease
MC	Whirling disease (<i>Myxobolus cerebralis</i>)
CS	<i>Ceratomyxa shasta</i>
ICH	Ichthyophthirius

Infectious Haematopoietic Necrosis

Infectious haematopoietic necrosis (IHN) is an infectious viral disease of rainbow or steelhead trout (*O. mykiss*), chinook (*O. tshawytscha*), sockeye (*O. nerka*), chum (*O. keta*), and coho (*O. kisutch*) salmon (OIE 2000). IHN reservoirs are clinically infected fish and covert carriers among cultured, feral, or wild fish. Once established in a farmed stock or watershed, due to either spawning of infected migratory fish or from river restocking for recreational purposes, IHN may become established among carriers (OIE 2000).

Table K.2 Potential impact of diseases in Pacific salmonids.

Pathogen/Parasite	Mortality	Reference
IHN Infectious hematopoietic necrosis	High in cultured and wild stocks; temperature-dependent	Bootland and Leong 2001, OIE 2000
IPN Infectious pancreatic necrosis	High (0–95%), especially in fry and fingerlings	Reno 1999, OIE 2000
VHS Viral hemorrhagic septicemia	High; temperature-dependent	McAllister 1990, OIE 2000
EIBS Erythrocytic inclusion body syndrome	Potentially high; however, co-occurs with other diseases	Piacentini et al. 1989
BKD Bacterial kidney disease	Low chronic to high acute epizootics at moderate temperatures	Bullock and Herman 1988, OIE 2000
FUR Furunculosis	High, especially at higher temperatures	Hiney and Olivier 1999
ERM Enteric redmouth disease	Sustained low- to large-scale, acute epizootics if stressed	Bullock and Cipriano 1990
CWD Cold-water disease	High (30–50%) in early fry stages; temperature-dependent	Shotts and Starliper 1999
PKD Proliferative kidney disease	Variable, but also temperature-dependent	Kinkelin and Lorient 2001
MC Whirling disease	High (up to 95%); temperature-dependent	Markiw 1992
CS <i>Ceratomyxa shasta</i>	High, especially if fish are previously unexposed	Bartholomew 1989, OIE 2000
ICH <i>Ichthyophthirius multifiliis</i>	High, especially if crowded; temperature-dependent	Traxler et al. 1998

Infectious Pancreatic Necrosis

Infectious pancreatic necrosis (IPN) is a highly contagious viral disease of young salmonids held under intensive rearing conditions. The disease most characteristically occurs in rainbow and steelhead trout, brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and several Pacific salmon species (OIE 2000). Transmission within a hatchery may be horizontal, vertical, or both (Reno 1999).

Viral Hemorrhagic Septicemia

Viral hemorrhagic septicemia (VHS), a systemic viral infection, occurs in salmonids of any age and may result in significant mortality (OIE 2000). Epizootic losses occur at temperatures of 3° to 12°C (greatest at 3° to 5°C); low daily mortality over an extended time results in high cumulative mortality. At high water temperatures (15° to 18°C), the disease acts quickly with modest accumulated mortality and fewer carriers (OIE 2000). Reservoirs of VHS are clinically infected fish and cultured, feral, or wild carriers, and animate or inanimate surfaces in hatcheries, where the virus can be mechanically transferred (McAllister 1990). Once established in stocks and water systems, carriers make the disease enzootic (OIE 2000).

Erythrocytic Body Inclusion Syndrome

Erythrocytic body inclusion syndrome (EIBS) is a viral condition observed in hatchery salmonids from the Columbia River and its tributaries (Piacentini et al. 1989). This syndrome occurs at a higher rate in hatchery chinook (70%) than those produced naturally (50%) in the Snake River Basin (PNWFHPC 1998). The disease is more severe in coho and chinook salmon than in rainbow and cutthroat trout and is more severe at higher temperatures (Piacentini et al. 1989).

Bacterial Kidney Disease

Bacterial kidney disease (BKD) is a systemic bacterial infection from *Renibacterium salmoninarum* that commonly causes high mortality in wild and propagated salmonids. The disease is typically chronic, but acute outbreaks sometimes occur at moderate temperatures (13° to 18°C), and subclinically infected fish or carriers are reservoirs of infection (Bullock and Herman 1988). Naturally infected feral brook trout can transmit BKD to newly stocked rainbow, brown, and brook trout, which begin dying within nine months. The bacteria are excreted by clinically diseased trout, and can survive up to 21 days in feces or pond sediments. BKD can also be transmitted vertically to eggs. Although health control measures may contain the disease, and different methods have been suggested for improving detection of the agent in infected fish populations, there is as yet no general agreement on the value of these methods (OIE 2000).

Furunculosis

Furunculosis (FUR) is a systemic bacterial infection that has an asymptomatic form, an acute form with severe internal organ involvement and high numbers of mortalities, and a chronic form that may involve internal pathology with fewer mortalities over longer periods of time (Reno 1999). Among salmonids, susceptibility is lower in brook, rainbow, and brown trout than in other species (Reno 1999).

Enteric Redmouth Disease

Enteric redmouth disease (ERM) is an acute or chronic bacterial infection from *Yersinia ruckeri*. Isolated from coho, sockeye, and chinook salmon and from rainbow, cutthroat, and brown trout, outbreaks have also been confirmed in steelhead (Horne and Barnes 1999). ERM commonly causes sustained, low-level mortality, eventually resulting in high losses; however, large-scale, acute epizootics occur if chronically infected fish are stressed via intensive culture and poor water quality (Bullock and Cipriano 1990, Horne and Barnes 1999). In rainbow trout, 25% of survivors of an experimental ERM challenge became asymptomatic carriers with bacteria localized in the lower intestine (Busch and Lingg 1975). In streams receiving hatchery effluent, 60% (3/5) of rainbow trout were positive for *Y. ruckeri* (Altinok et al. 2001). While commercial vaccines have been effective, strains have developed resistance and questions remain about the nature of antigens involved (Horne and Barnes 1999).

Cold-Water Disease

Cold-water disease (CWD) is a bacterial infection that occurs in salmon and brook, rainbow, and brown trout. Morbidity ranges from 1% to 50%; at low temperatures (below 10°C), it approaches 75% (Shotts and Starliper 1999). Transmission is vertical or horizontal, and outbreaks can occur after stocking a habitat with infected fish or transferring fish from facilities where the infection had not been detected.

Proliferative Kidney Disease

Proliferative kidney disease (PKD) is caused by a parasitic myxozoan infection that also parasitizes bryozoans (Kinkelin and Boriot 2001). Mortality from PKD is variable and temperature dependent.

Whirling Disease

Whirling disease (MC) is a parasitic infection of trout and salmon by the myxosporean *Myxobolus cerebralis*. Susceptibility ranges from high to low in rainbow trout, sockeye salmon, golden trout (*O. aguabonita*), cutthroat trout, brook trout, steelhead, chinook salmon, brown trout, and coho salmon. Susceptibility is greater in younger fish than in older fish (Markiw 1992). The source of the infective agent for fish is usually the water supply or earthen ponds

inhabited by aquatic tubificid worms. Mortalities up to 90% may occur between newly hatched fish exposed to the infective agent as sac fry.

Ceratomyxa shasta

Ceratomyxa shasta (CS) is a parasite that leads to mortality of hatchery-reared and wild juvenile salmonids as well as to pre-spawning mortality in adult salmon (Bartholomew 1989). Documented in chinook, coho, sockeye, and chum salmon, as well as steelhead, rainbow, brook, brown, and cutthroat trout, it involves an intermediate host, the polychaete worm *Manaynukia speciosa*. Control of the parasite in hatchery and wild populations depends on the introduction of resistant salmonids (Bartholomew 1989), thus epizootics are possible if infested water is transferred to native populations.

Ichthyophthirius

Ichthyophthirius (ICH), or "white spot disease," is a protozoan infection of freshwater fish caused by *Ichthyophthirius multifiliis*. The parasite is quite lethal, and epizootics occur with relative predictability. As the free-swimming infective stage is viable for only days, epizootics are more likely to occur in facilities with high-density fish populations. The "Ich" life cycle is influenced by crowding, and growth rate and development accelerate when water temperatures reach between 16° and 19°C (Traxler et al. 1998).

While data on occurrence and impact of hatchery diseases provide information on historical potential for catastrophic epidemics, continuing high production of hatchery fish may increase the risk of future epidemics despite hatchery practices that may have been instituted to mitigate or eliminate mortality in hatcheries. The densities at which hatchery populations are reared and released may increase the potential for known or heretofore unreported diseases to spread within hatchery populations and then from hatchery populations to wild populations. In dense populations, pathogen incidence can be high and transmission barriers can be low, which can lead to high virulence. Dense hatchery populations may act as reservoirs for exotic pathogens, and, if hatchery fish are asymptomatic, transmission to wild populations can be accelerated.

Pollution

Pollution in the form of oil and chemical spills can pose a risk to salmonid populations in the Lower Columbia and Upper Willamette ESU. Fish kills have been reported associated with such spills, and the frequency of these events has not necessarily been reduced by prophylactic measures and legislated improvements over the past few decades. There are point sources of spills and leaks at storage facilities and superfund sites as well as from the transportation of oil and chemicals.

In addition to the myriad anthropogenic factors that can affect the survival of outmigrating juveniles in freshwater, the biological effects of chemical contaminants on salmonids during their residency in certain urban estuaries can potentially lead to reduced

survival. Concomitant with the increased chemical exposure, juvenile salmonids inhabiting certain urban estuaries exhibit evidence of impairment of physiological processes such as immune system alterations, impaired growth, and behavioral changes. There is evidence of linkage between the presence of elevated levels of complex mixtures of chemical contaminants in polluted estuaries and effects on health and survival of juvenile salmonids. Sublethal effects from toxic chemical exposure experienced by outmigrant juvenile salmonids during their residence in urbanized estuaries indicate the need to further investigate estuarine pollution as a contributing factor to declines in salmon stocks from urbanized watersheds (Casillas et al. 1997).

Methods

Volcanoes

To determine relative risk from volcanic activity, we assembled hazard assessment data from U.S. Geological Survey (USGS) sources for Mount Rainier (Hoblitt et al. 1998), Mount Adams (Scott et al. 1995), Mount St. Helens (Wolfe and Pierson 1995), Mount Hood (Scott et al. 1997), Mount Jefferson (Walder et al. 1999), and the Three Sisters region (Scott et al. 2001). In these reports, areas around volcanoes are divided into proximal and distal hazard zones—some zones are subdivided further—based on magnitude of past volcanic events inferred from deposits, mathematical models that use calibrations from other volcanoes to forecast the likely extent of future pyroclastic flows, debris avalanches and lahars, and experience and judgment of USGS scientists derived from observations and understanding of events at similar volcanoes.

Hazard assessments were overlaid, along with fish distributions, on maps of populations of the five Lower Columbia River and Upper Willamette River salmonid ESUs. The relative catastrophic risk among populations was assessed for Lower Columbia fall and spring chinook, Upper Willamette spring chinook, Lower Columbia winter and summer steelhead, Upper Willamette winter steelhead, and Lower Columbia chum by categorizing volcanic hazard for each population. Categories of relative catastrophic risk from volcanoes included negligible, low, medium, and high. Populations that did not overlap with any volcanic hazard zone were categorized as negligible. Populations that overlapped with low hazard zones were categorized as low. Populations that overlapped with low and medium volcanic hazard zones were categorized as medium. Populations that overlapped with medium and high volcanic hazard zones were categorized as high.

Earthquakes

To determine relative risk from earthquake activity, we assembled hazard assessment data for the Pacific Northwest from USGS report 97-131 (Frankel et al. 1997). Hazard probabilities were calculated from the largest ground motions to the smallest at a collection of sites and added up to a total probability, P , and in a particular period of time, T . For our analyses, we chose the hazard assessment of rare, large-scale events. The hazard contours in this analysis are represented as peak acceleration (% gravity [g]) or the percent acceleration force due to gravity with a 2% probability of exceedance (PE) in 50 years.

Hazard probabilities were overlaid with fish distributions on populations of the five Lower Columbia River and Upper Willamette River salmonid ESUs. The relative catastrophic risk among populations was assessed for Lower Columbia fall and spring chinook, Upper Willamette spring chinook, Lower Columbia winter and summer steelhead, Upper Willamette winter steelhead, and Lower Columbia chum by categorizing earthquake hazard for each population. Categories of relative catastrophic risk from earthquakes included low, medium, and high. Populations that overlapped with earthquake hazard zones with low peak acceleration values (18%–30% g) were categorized as low. Populations that overlapped with earthquake hazard zones with moderate peak acceleration values (30%–60% g) were categorized as medium. Populations that overlapped with earthquake hazard zones with high peak acceleration values (60%–120%g) were categorized as high.

Landslides/Glacial Outbursts

To determine relative risk from landslide activity, we assembled hazard assessment data from USGS sources (Godt 1997). The map was trimmed to the Lower Columbia River/Upper Willamette River ESU boundaries and overlaid with major rivers on populations of the five Lower Columbia River and Upper Willamette River salmonid ESUs. The relative catastrophic risk among populations was assessed for Lower Columbia fall and spring chinook, Upper Willamette spring chinook, Lower Columbia winter and summer steelhead, Upper Willamette winter steelhead, and Lower Columbia chum by categorizing landslide hazards for each population. The original USGS coverage categorized risk in two ways: landslide susceptibility/incidence and landslide incidence. We grouped these assessments into catastrophic risk categories of low, medium, and high based on the relative amounts of area in each population in the three categories and the proximity of hazard areas to salmonid-bearing streams in the population. Populations that overlapped with landslide hazard zones of low hazard values were categorized as low. Populations that overlapped with medium values or a combination of hazard zones that averaged medium were categorized as medium. Populations that overlapped with landslide hazard zones with high values or a combination of hazard zones that averaged or was dominated by high values, especially in fish-bearing streams, was categorized as high.

Disease Epidemics from Hatchery Operations

As a proxy for the potential for disease epidemics, we collected data on the extent of hatchery production within population boundaries for the five Lower Columbia River and Upper Willamette River salmonid ESUs. Data on hatchery production levels were compiled from a variety of sources, primarily the 2000 hatchery adipose-clip database (<ftp://ftp.streamnet.org/pub/streamnet/maps/massmarking>), hatchery genetic management plans submitted to NMFS, Integrated Hatchery Operations Team (IHOT) reports (www.streamnet.org/ihot_audit/hatchery.html), and the Northwest Indian Fisheries Commission Hatchery Releases Web page (www.NWIFC.org). Annual releases of salmonids (chinook, chum, coho, sockeye, steelhead, and cutthroat trout) and related species (rainbow, brown, brook, and golden trout) at on-site and at off-site locations were summed for facility and categorized as no risk (0 fish/year), low risk (0 to 500,000 fish/year), medium risk (500,000 to 5,000,000 fish/year) and high risk (>5,000,000

fish/year). Population boundaries were defined according to historical demographically independent populations identified in Myers et al. (2002).

Oil/Chemical Pollution (Transportation)

To determine relative risk of a catastrophic event due to oil/chemical pollution, we assembled information on transportation corridors in areas overlapping with the listed ESUs in the Lower Columbia and Upper Willamette Rivers. The road density GIS layer was obtained from the Regional Ecosystem Office Web site (<http://www.reo.gov/reo/>) and overlaid, along with fish distributions, on maps of populations of the five Lower Columbia and Upper Willamette River salmonid ESUs. A ratio of road density was calculated using ArcView 8.1 by dividing the linear extent (km) of all roads by the total area (km²) encompassed by each population. We mapped the ratio of road length/area of population (see Figures K.17–K.22) based on sorting the ratios into four equal intervals and labeled them accordingly: negligible, low, medium, and high. The relative catastrophic risk among populations was assessed for ESUs (Lower Columbia chum) and life history types within ESUs (Lower Columbia fall and spring chinook; Upper Willamette spring chinook; Lower Columbia winter and summer steelhead; Upper Willamette winter steelhead) by categorizing earthquake hazard for each population.

Correlated Catastrophic Risk Assessment

Single catastrophic events can affect a single population or an entire metapopulation. For Pacific salmon metapopulations, an intermediate case is appropriate, whereby a single catastrophic event will affect several populations, but not necessarily the entire metapopulation. For example, a volcanic eruption could drastically reduce spawning and rearing habitat for populations in multiple watersheds. It is possible to explore how these types of spatially correlated catastrophes affect metapopulation dynamics by simulating the effect of catastrophes on population-specific capacities within an ESU. Given a hypothesized spatial correlation in risk between 21 chinook populations of the Puget Sound ESU, the program RAMAS 4.0 applied catastrophes stochastically over 100 years, with a per-population catastrophic risk that was increased according to its correlation with other populations. For this ESU, catastrophic events can affect the performance of a metapopulation (Ruckelshaus et al. in prep.).

Results of Analyses

Volcanoes

The catastrophic risk from volcanic activity varied among populations within ESUs, but generally depended upon proximity to the north-south line of Cascade volcanoes. The populations and their risks are as follows:

Lower Columbia River fall chinook ESU (Figure K.1)

Negligible (9) Coast Range—Youngs Bay, Grays River, Big Creek, Elochoman River, Clatskanie River, Mill Creek, Scappoose Creek

	Western Cascades—Coweeman and Clackamas River tributaries
Low (5)	Western Cascades—upper and lower Cowlitz River, Washougal River Columbia Gorge—lower and upper gorge tributaries
Medium (3)	Western Cascades—Lewis River/Salmon Creek Columbia Gorge—Big White Salmon River, Hood River tributaries
High (3)	Western Cascades—Toutle, Kalama, Sandy Rivers
Lower Columbia River spring chinook ESU (Figure K.2)	
Negligible (1)	Western Cascades—Tilton River
Medium (5)	Western Cascades—upper Cowlitz, Cispus, and Lewis Rivers Columbia Gorge—Big White Salmon and Hood Rivers
High (3)	Western Cascades—Toutle, Kalama, and Sandy Rivers
Upper Willamette spring chinook ESU (Figure K.2)	
Negligible (3)	Molalla, South Santiam, and Calapooia Rivers
Low (2)	Clackamas and Middle Fork Willamette Rivers
Medium (2)	North Santiam and McKenzie Rivers
Lower Columbia winter steelhead ESU (Figure K.3)	
Negligible (4)	Western Cascades—Tilton, Coweeman, and East Fork Lewis Rivers, and Salmon Creek
Low (5)	Western Cascades—lower Cowlitz, Clackamas, and Washougal Rivers Columbia Gorge—lower and upper gorge tributaries.
Medium (4)	Western Cascades—Cispus, upper Cowlitz, and North Fork Lewis Rivers. Columbia Gorge—Hood River
High (4)	Western Cascades—North and South Forks Toutle, Kalama, and Sandy Rivers
Lower Columbia River summer steelhead ESU (Figure K.4)	
Negligible (1)	Western Cascades—East Fork Lewis River
Low (2)	Western Cascades—Washougal and Wind Rivers
Medium (2)	Western Cascades—North Fork Lewis River. Columbia Gorge—Hood River
High (1)	Western Cascades—Kalama River
Upper Willamette River winter steelhead ESU (Figure K.3)	
Negligible (4)	Willamette Valley—Coast Range tributaries, Molalla River, South Santiam River, and Calapooia River
Medium (1)	Willamette Valley—North Santiam River
Lower Columbia River chum ESU (Figure K.5)	
Negligible (8)	Coast Range—Youngs Bay, Grays River (including Chinook River), Big Creek, Elochoman River, Clatskanie River, Mill Creek, Scappoose Creek Western Cascades—Salmon Creek
Low (5)	Western Cascades—lower Cowlitz, Clackamas, and Washougal Rivers Columbia Gorge—lower and upper gorge tributaries

Medium (1)	Western Cascades—Lewis River
High (2)	Western Cascades—Kalama River, Sandy River

Correlated Catastrophic Risk from Volcanoes

The catastrophic risk posed by volcanic activity transcends population boundaries delineated for many ESUs. Some volcanoes present a clear and present danger for many ESUs and populations therein, and some populations were under hazard from multiple volcanoes. Because of the spatial arrangement of the Cascade Mountain volcanoes, correlated catastrophic risks are not necessarily reciprocal, especially where tributaries form distinct populations. For these reasons, as well as the categorical nature of the risk assessments, we did not construct a quantitative correlated catastrophic risk matrix. Simultaneous catastrophic risk to ESU populations, by volcano, are as follows:

Lower Columbia River fall chinook ESU

- Mount St. Helens—Lewis River/Salmon Creek, Kalama and Toutle Rivers, lower Cowlitz River population downstream of the Toutle
- Mount Rainier—upper and lower Cowlitz populations
- Mount Adams—Washougal River, Big White Salmon River, and lower and upper gorge tributaries
- Mount Hood—Hood and Sandy Rivers

Lower Columbia River spring chinook ESU

- Mount St. Helens—Toutle and Kalama Rivers
- Mount Rainier—upper Cowlitz River, but not the Cispus River population
- Mount Adams—Cispus River, upper Cowlitz River via the Cispus, and Big White Salmon River
- Mount Hood—Hood and Sandy Rivers

Upper Willamette spring chinook ESU

- Mount Jefferson—Clackamas and North Santiam Rivers
- Three Sisters—McKenzie and Middle Fork Willamette Rivers

Lower Columbia winter steelhead ESU

- Mount St. Helens—North and South Fork Toutle, Kalama, and North Fork Lewis Rivers
- Mount Rainier—upper and lower Cowlitz Rivers
- Mount Adams—Cispus River, upper and lower Cowlitz River via the Cispus River, and Washougal River, lower and upper gorge tributaries
- Mount Hood—Hood and Sandy Rivers

Lower Columbia River summer steelhead ESU

- Mount St. Helens—Kalama River and North Fork Lewis River
- Mount Adams—North Fork Lewis, Washougal, and Wind Rivers

Upper Willamette River winter steelhead ESU

- Mount St. Helens—North and South Fork Toutle, Kalama, and North Fork Lewis Rivers
- Mount Rainier—upper and lower Cowlitz River
- Mount Adams—Cispus River, upper and lower Cowlitz River via the Cispus, Washougal River, and lower and upper gorge tributaries
- Mount Hood—Hood and Sandy Rivers

Lower Columbia River chum ESU

- Mount St. Helens—lower Cowlitz, Kalama, and Lewis Rivers
- Mount Adams—Washougal River, lower and upper gorge tributaries
- Mount Hood—Hood and Sandy Rivers

Earthquakes

The catastrophic risk from earthquakes varied among populations within ESUs but generally declined from coastal to inland tributaries.

Lower Columbia River fall chinook (Figure K.6)

- | | |
|-------------|---|
| Low (5) | Western Cascades—Sandy River
Columbia Gorge—lower and upper gorge tributaries, Big White Salmon River, Hood River |
| Medium (13) | Coast Range—Grays River, Big Creek, Elochoman River, Clatskanie River, Mill Creek, Scappoose Creek
Western Cascades—Cowlitz, Coweeman, Toutle, and Kalama Rivers, Lewis River/Salmon Creek, Clackamas and Washougal Rivers |
| High (1) | Coast Range—Youngs Bay |

Lower Columbia River spring chinook (Figure K.7)

- | | |
|------------|---|
| Low (3) | Western Cascades—(Sandy River)
Columbia Gorge—Big White Salmon River, Hood River |
| Medium (6) | Western Cascades—upper Cowlitz, Cispus, Tilton, Toutle, Kalama, and Lewis Rivers |

Upper Willamette River spring chinook (Figure K.7)

- | | |
|------------|---|
| Low (6) | Willamette Valley and western Cascades—Clackamas, North and South Santiam, Calapooia, McKenzie, and Middle Fork Willamette Rivers |
| Medium (1) | Willamette Valley—Molalla River |

Lower Columbia River winter steelhead (Figure K.8)

- | | |
|-------------|--|
| Low (5) | Western Cascades—Clackamas and Sandy Rivers
Columbia Gorge—lower and upper gorge tributaries and Hood River |
| Medium (12) | Western Cascades—Cispus, Tilton, lower and upper Cowlitz, North and South Fork Toutle, Coweeman, Kalama, North and East Fork Lewis Rivers, Salmon Creek, and Washougal River |

Lower Columbia River summer steelhead (Figure K.9)

- Low (2) Columbia Gorge—Hood and Wind Rivers
- Medium (4) Western Cascades—Kalama, North and East Fork Lewis, and Washougal Rivers

Lower Columbia River winter steelhead (Figure K.8)

- Low (3) Willamette Valley and western Cascades—North and South Santiam and Calapooia Rivers
- Medium (2) Willamette Valley and western Cascades—Coast Range tributaries and Molalla River

Lower Columbia River chum (Figure K.10)

- Low (2) Columbia Gorge—lower and upper gorge tributaries
- Medium (13) Coast Range—Grays River (including Chinook River), Big Creek, Elochoman River, Clatskanie River, Mill Creek, Scappoose Creek
Western Cascades—lower Cowlitz River, Kalama River, Salmon Creek, Lewis River, Clackamas River, Washougal River, Sandy River
Coast Range—Youngs Bay

Correlated Catastrophic Risk from Earthquakes

The catastrophic risk posed by earthquake activity transcends population boundaries delineated for many ESUs and tends to vary along a gradient from the coast eastward toward the interior. Populations within ESUs are under the same level of hazard risk along this east-west gradient; thus, populations within ecoregions (Coast Range, western Cascades, Columbia Gorge) tend to be under correlated risk from catastrophic earthquake activity. For these reasons, as well as the categorical nature of the risk assessments, we did not construct a quantitative correlated catastrophic risk matrix. Simultaneous catastrophic risk to ESU populations by earthquakes are as follows:

Lower Columbia River fall chinook

- Coast Range (7)—Youngs Bay, Grays River, Big Creek, and Elochoman River,
Clatskanie River, Mill Creek, and Scappoose Creek
- Western Cascades (9)
- Columbia Gorge (4)

Lower Columbia River spring chinook ESU

- Western Cascades (7)
- Columbia Gorge (2)

Upper Willamette spring chinook

- Willamette Valley/Western Cascades (7)

Lower Columbia winter steelhead

- Western Cascades (14)

Columbia Gorge (3)

Lower Columbia River summer steelhead (6)

Western Cascades (4)

Columbia Gorge (2)

Upper Willamette River winter steelhead (5)

Coast Range—Molalla, Santiam, South Santiam, and Calapooia Rivers

Lower Columbia River chum

Coast Range (7)—Youngs Bay, Grays River (including Chinook River), and Big Creek, Elochoman River, Clatskanie River, Mill Creek, and Scappoose Creek

Western Cascades (9)

Columbia Gorge (2)

Landslides

The catastrophic risk from landslides varied among populations for populations within ESUs.

Lower Columbia River fall chinook (Figure K.11)

Low (5) Western Cascades—Upper Cowlitz River, Coweeman River, Lewis River /Salmon Creek

Columbia Gorge—Big White Salmon River, Hood River

Medium (5) Coast Range—Grays River, Elochoman River, Mill Creek

Western Cascades—Toutle and Washougal Rivers

High (10) Coast Range—Youngs Bay, Big Creek, Clatskanie River, Scappoose Creek

Western Cascades—Lower Cowlitz, Kalama, Clackamas, and Sandy Rivers

Columbia Gorge—lower and upper gorge tributaries

Lower Columbia River spring chinook (Figure K.12)

Low (4) Western Cascades—Cispus and Tilton Rivers

Columbia Gorge—Big White Salmon and Hood Rivers

Medium (3) Western Cascades—upper Cowlitz, Toutle, and Lewis Rivers

High (2) Western Cascades—Kalama and Sandy River

Upper Willamette River spring chinook (Figure K.12)

Low (1) Willamette Valley and western Cascades—Molalla River

Medium (3) Willamette Valley and western Cascades—Clackamas, McKenzie, and Middle Fork Willamette Rivers

High (3) Willamette Valley and western Cascades—North and South Santiam and Calapooia Rivers

Lower Columbia River winter steelhead (Figure K.13)

- Low (6) Western Cascades—Cispus, Tilton, South Fork Toutle, Coweeman, and East Fork Lewis Rivers and Salmon Creek
- Medium (6) Western Cascades—upper Cowlitz, North Fork Toutle, North Fork Lewis, Clackamas, and Washougal Rivers
Columbia Gorge—Hood River
- High (5) Western Cascades—lower Cowlitz, Kalama, and Sandy River
Columbia Gorge—lower and upper gorge tributaries

Lower Columbia River winter steelhead (Figure K.14)

- Low (3) Western Cascades—East Fork Lewis River
Columbia Gorge—Wind and Hood Rivers
- Medium (1) Western Cascades—North Fork Lewis River
- High (2) Western Cascades—Kalama and Washougal Rivers

Lower Columbia River winter steelhead (Figure K.13)

- Low (1) Molalla River
- Medium (1) Coast Range tributaries
- High (3) North and South Santiam and Calapooia Rivers

Lower Columbia River chum ESU (Figure K.15)

- Low (2) Salmon Creek, Lewis River
- Medium (3) Western Cascades—Grays River, Mill Creek, Washougal River
- High (11) Coast Range—Youngs Bay, Big Creek, Elochoman River, Clatskanie River, Scappoose Creek
Western Cascades—lower Cowlitz, Kalama, Clackamas, and Sandy Rivers
Columbia Gorge—lower and upper gorge tributaries

Correlated Catastrophic Risk from Landslides

The catastrophic risk posed by landslides transcends population boundaries delineated for many ESUs and is highly variable across the landscape. The catastrophic risk posed by landslide activity is highly influenced by several factors, including a variety of geologic factors and precipitation patterns, thus the few areas at a high risk from landslides may or may not be along salmonid spawning or rearing habitat. Landslide risk due to earthquake or volcanic activity will mirror those assessments, while those associated with flooding and precipitation will mirror those assessments. For these reasons, as well as the categorical nature of the risk assessments, we did not construct a quantitative correlated catastrophic risk matrix.

Disease Epidemics from Hatchery Operations

The potential for disease epidemics as represented by hatchery production varied among populations within ESUs and individual hatchery facilities.

Lower Columbia River fall chinook ESU (Figure K.16)

- Negligible (4) Coast Range—Clatskanie River, Mill Creek, Scappoose Creek
Western Cascades—Coweeman River
- Low (4) Western Cascades—Salmon Creek
Columbia Gorge—Big White Salmon and Hood Rivers
- Medium (5) Coast Range—Youngs Bay, Grays River
Western Cascades—Toutle, Clackamas, and Sandy Rivers
- High (8) Coast Range—Big Creek, Elochoman River
Western Cascades—Cowlitz, Kalama, Lewis, and Washougal Rivers
Columbia Gorge—lower and upper gorge tributaries

Lower Columbia River spring chinook ESU (Figure K.17)

- Negligible (3) Western Cascades—upper Cowlitz, Cispus, and Tilton Rivers
- Low (2) Columbia Gorge—Big White Salmon and Hood Rivers
- Medium (2) Western Cascades—Toutle and Sandy Rivers
- High (2) Western Cascades—Kalama and Lewis Rivers

Upper Willamette River spring chinook ESU (Figure K.17)

- Negligible (2) Molalla and Calapooia Rivers
- Medium (4) Clackamas, North Santiam, McKenzie, and Middle Fork Willamette Rivers
- High (1) South Santiam River

Lower Columbia River winter steelhead ESU (Figure K.18)

- Negligible (6) Western Cascades—Cispus, Tilton, upper Cowlitz, South Fork Toutle, Coweeman, and East Fork Lewis Rivers
- Low (2) Western Cascades—Salmon Creek
Columbia Gorge—Hood River
- Medium (3) Western Cascades—North Fork Toutle, Clackamas, and Sandy Rivers
- High (6) Western Cascades—lower Cowlitz, Kalama, North Fork Lewis, and Washougal Rivers
Columbia Gorge—lower and upper gorge tributaries, Hood River

Upper Willamette River winter steelhead ESU (Figure K.18)

- Negligible (2) Westside (Coast Range)—Molalla and Calapooia Rivers
- Medium (1) North Santiam River
- High (1) South Santiam River

Lower Columbia River summer steelhead ESU (Figure K.19)

- Negligible (1) Western Cascades—East Fork Lewis River
- Low (1) Columbia Gorge—Hood River
- Medium (1) Columbia Gorge—Wind River
- High (3) Western Cascades—Kalama, North Fork Lewis, and Washougal Rivers

Lower Columbia River chum ESU (Figure K.20)

- Negligible Coast Range—Clatskanie River, Mill Creek, Scappoose Creek

	Western Cascades—Coweeman River
Low (1)	Western Cascades—Salmon Creek
Medium (4)	Coast Range—Youngs Bay, Grays River
	Western Cascades—Clackamas and Sandy Rivers
High (7)	Coast Range—Big Creek, Elochoman River
	Western Cascades—lower Cowlitz, Kalama, Lewis, and Washougal Rivers
	Columbia Gorge—lower and upper gorge tributaries

Correlated Catastrophic Risk from Hatchery Disease Epidemics

Correlated catastrophes were considered for hatchery disease epidemics in populations within ESUs. The connection among populations in terms of correlated catastrophes depends on many factors. If infected fish were released into the wild to prevent die-offs at the hatchery, as has occurred in the past, potential infection of wild fish would be a function of infected hatchery fish and wild fish densities as well as of pathogen or parasite transmission rates. Further, the spread of infections up or downstream would depend on pathogen or parasite movement patterns as well as wild fish movement at the time of potential transmission. For these reasons, we did not construct a quantitative correlated catastrophic risk matrix.

Pollution

The catastrophic risk of oil/chemical pollution from transportation varied among populations within ESUs, but generally followed patterns of urban development throughout the river basins.

Lower Columbia River fall chinook ESU (Figure K.21)

Negligible (8)	Coast Range—Big Creek
	Western Cascades—upper and lower Cowlitz, Washougal, and Sandy Rivers
	Columbia Gorge—lower and upper gorge tributaries, Big White Salmon River
Low (8)	Coast Range—Youngs Bay, Grays River, Elochoman River, Clatskanie River, Scappoose Creek
	Western Cascades—Toutle River, Lewis River/Salmon Creek
	Columbia Gorge—Hood River
Medium (3)	Coast Range—Mill Creek
	Western Cascades—Coweeman and Kalama River
High (1)	Western Cascades—Clackamas River

Lower Columbia River spring chinook ESU (Figure K.22)

Negligible (4)	Western Cascades—upper Cowlitz, Cispus, and Sandy Rivers
	Columbia Gorge—Big White Salmon River
Low (2)	Western Cascades—Lewis River
	Columbia Gorge—Hood River

Medium (2)	Western Cascades—Tilton and Toutle Rivers
High (2)	Western Cascades—Kalama River
Upper Willamette River spring chinook ESU (Figure K.23)	
Negligible (2)	McKenzie and Middle Fork Willamette Rivers
Low (3)	Clackamas, North Santiam, and Calapooia Rivers
High (2)	Molalla and South Santiam Rivers
Lower Columbia River winter steelhead ESU (Figure K.24)	
Negligible (4)	Western Cascades—Cispus, upper Cowlitz, and Sandy Rivers Columbia Gorge—lower gorge tributaries
Low (4)	Western Cascades—North and East Forks Lewis River Columbia Gorge—upper gorge tributaries and Hood River
Medium (5)	Western Cascades—Tilton, lower Cowlitz, North Fork Toutle, Clackamas, and Washougal Rivers
High (4)	Western Cascades—South Fork Toutle, Coweeman, and Kalama Rivers and Salmon Creek
Lower Columbia River summer steelhead ESU (Figure K.25)	
Negligible (3)	Western Cascades—North Fork Lewis and Washougal Rivers Columbia Gorge—Wind River
Low (2)	Western Cascades—East Fork Lewis River Columbia Gorge—Hood River
High (1)	Western Cascades—Kalama River
Upper Willamette River winter steelhead ESU (Figure K.26)	
Negligible	North Santiam and Calapooia Rivers
High (3)	Western (Coast Range) tributaries and Molalla and South Santiam Rivers
Lower Columbia River chum populations ESU (Figure K.27)	
Negligible	Coast Range—Big Creek Columbia Gorge—lower and upper gorge tributaries
Low	Coast Range—Youngs Bay, Elochoman River, Clatskanie River, Scappoose Creek Western Cascades—Lewis, Washougal, and Sandy Rivers
Medium (5)	Coast Range—Grays River, Mill Creek Western Cascades—lower Cowlitz River, Kalama River, Salmon Creek
High (1)	Western Cascades—Clackamas River

Correlated Catastrophic Risk from Oil/Chemical Spills from Transportation

Correlated catastrophes were considered for oil/chemical spills from transportation among populations in ESUs. Like for hatchery disease, however, the connection among populations in terms of correlated catastrophes would depend on many factors. The oil/chemical spills likely to occur during transportation over roadways would probably be confined within a

watershed, and direct mortality from toxins would be on a subpopulation scale. However, spills from roadways into a tributary might affect its major river; spread of the mortality agent upstream or downstream would depend on volatility of the oil/chemical as well as movement of wild fish at the time of the potential transmission. For these reasons, we did not construct a quantitative correlated catastrophic risk matrix.

Conclusions

Catastrophic events need to be considered when relating viable salmonid populations (VSPs) to viable ESUs in the Lower Columbia and Willamette Rivers. Although documenting the frequency, intensity, and hazard risk of specific natural and anthropogenic catastrophes is possible across the landscape for ESUs, calculating correlated catastrophic risk can be problematic for some catastrophes. Harder still is the task of calculating cumulative effects of volcanoes, earthquakes, floods, landslides, fires, disease epidemics from hatcheries, and pollution from a variety of sources. Still, the preponderance of potential catastrophic events that could impact salmonids throughout the Columbia River Basin requires attention to their potential effects, and the paucity of such approaches belies its importance. The potential catastrophes cataloged herein represent an initial list of those where (1) the risks for salmonids have been documented or are known to represent a future risk, (2) actual risk information (or a reasonable proxy) has been collated or is accessible, and (3) there is potential for quantitative data in the future. Further analyses of catastrophic risks are ongoing (floods) or may be initiated (extreme weather such as droughts, unusual fires, water diversion/dam failure, major miscalculations in harvest) depending on the information available and the potential for rigorous analyses. More refined metrics may allow for further exploration of the risk of disease epidemics from hatchery operations.

The role that catastrophes may have played in the evolution of salmonids suggests that "bet-hedging" against large-scale catastrophes through maintaining diverse populations and life-history types is an appropriate strategy for recovery in the face of extinction risk. Such a strategy should foster enhanced long-term stability in the face of unpredictable catastrophes. Future research on the risk of extinction posed by catastrophic events for an entire ESU will hinge on quantitative estimates of correlated risk among populations within an ESU. Guidelines presented in McElhany et al. (2000) make clear that concern about catastrophic risks is relevant to long-term evolutionary potential. The probability that an ESU could be driven extinct by a single catastrophic event is nontrivial and thus requires multiple viable populations within a viable ESU, with careful consideration to which populations are restored or maintained at viable status.

This appendix explored the spatial distribution and frequencies of potential natural and anthropogenic catastrophic events affecting endangered Pacific salmonid ESUs in the Lower Columbia and Upper Willamette Rivers, specifically chinook salmon (*Oncorhynchus tshawytscha*), steelhead trout (*O. mykiss*), and chum salmon (*O. keta*). While this is a difficult field of study, we conclude that extinction risk, particularly with respect to catastrophic events, can be reduced if viable populations are spatially distributed through out the ESU. Spatially distributed populations utilizing different environments with different catastrophic risks reduce the likelihood that a single catastrophic event would affect every population in an ESU. Further, fish with different life histories that share the same river basin may be affected differentially by

the same catastrophic event. This spreading of risk throughout spatially distributed populations and life-history strata, akin to the “bet-hedging” that occurred during the evolution of salmonids, holds promise for reducing the risk of extinction due to catastrophes for these endangered and threatened Pacific salmonid ESUs in the Lower Columbia and Upper Willamette Rivers.

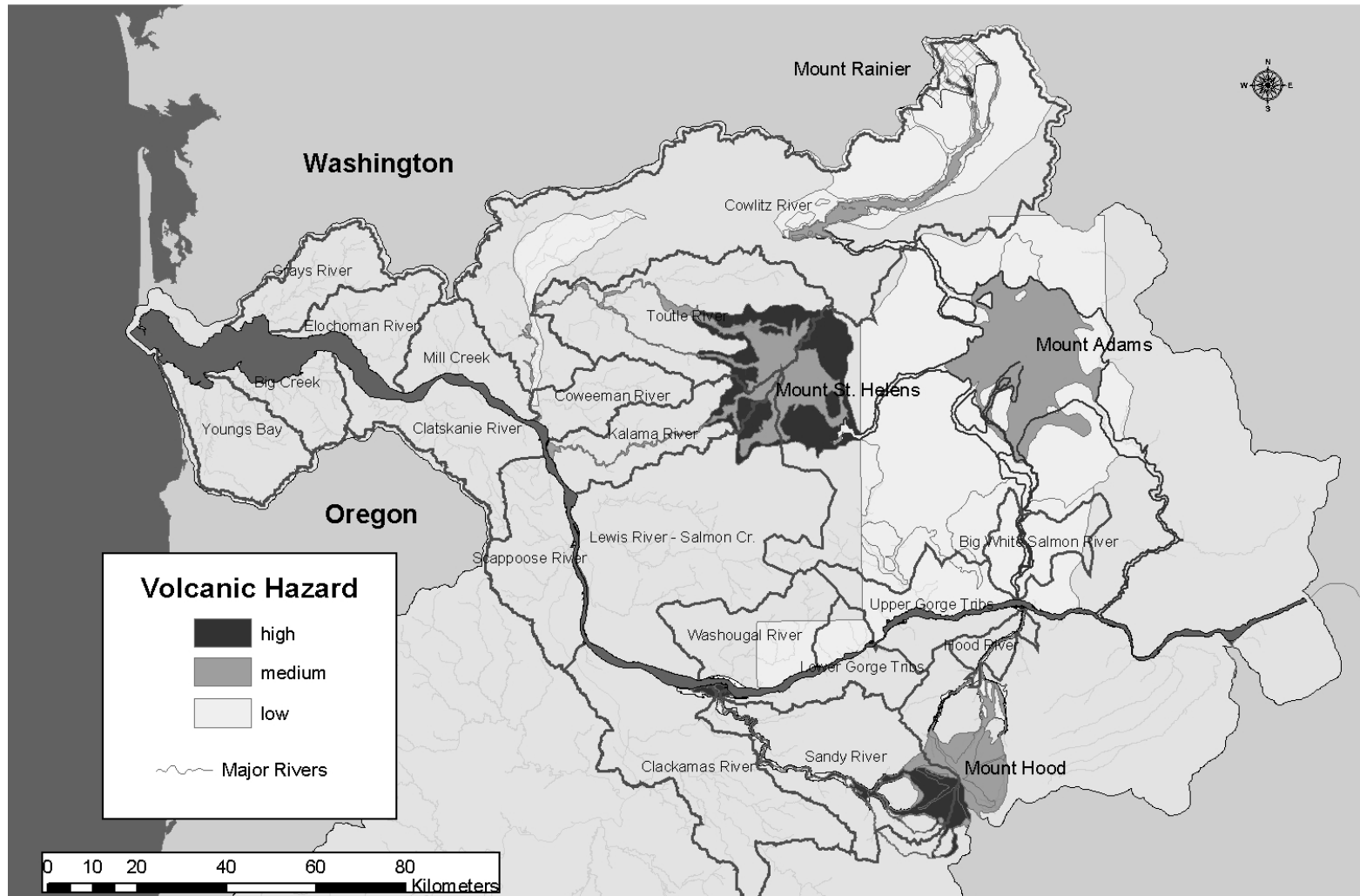


Figure K.1 Fall chinook—Lower Columbia ESU volcanic hazards.

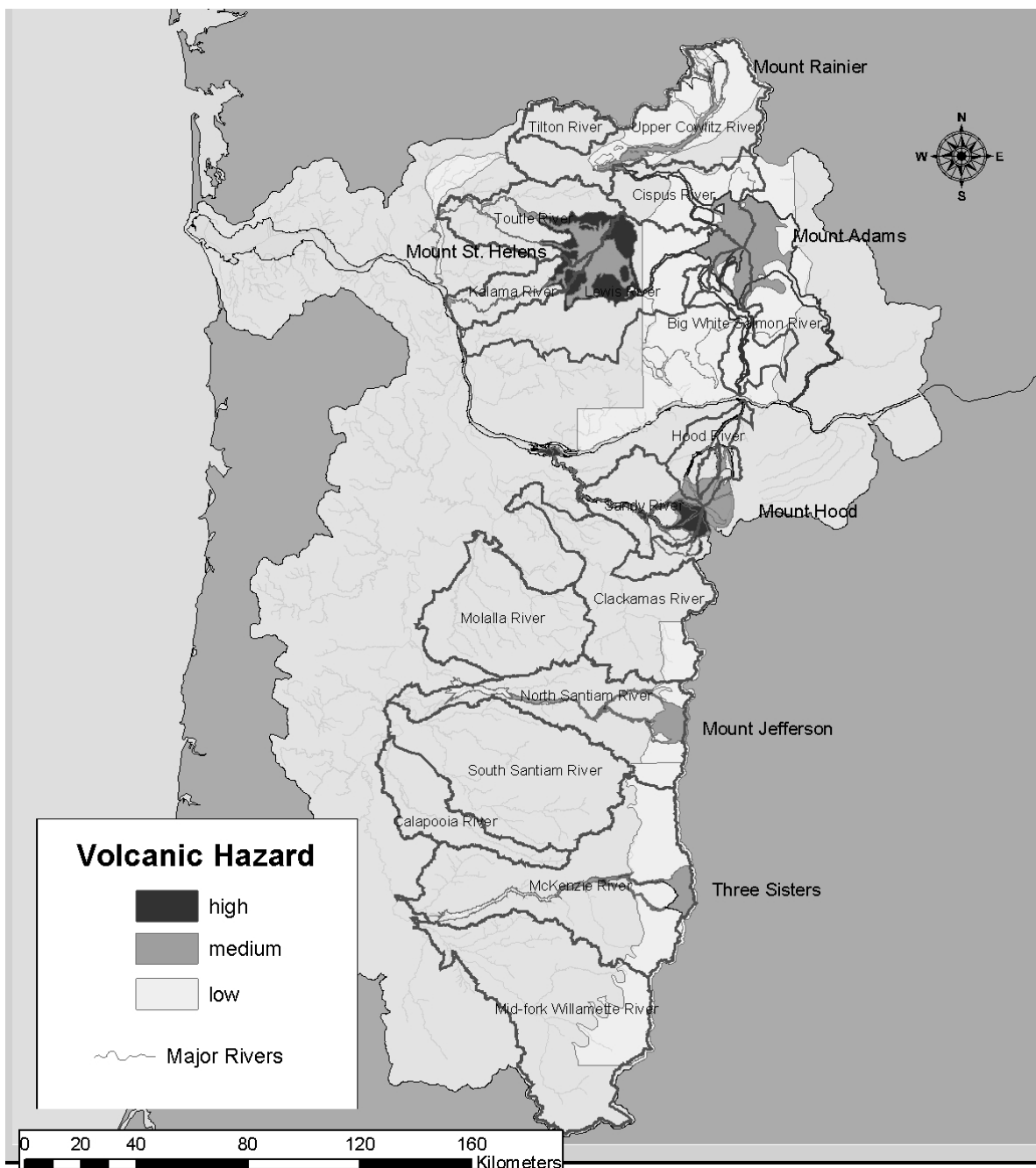


Figure K.2 Spring chinook—Lower Columbia and Willamette River ESUs volcanic hazards.

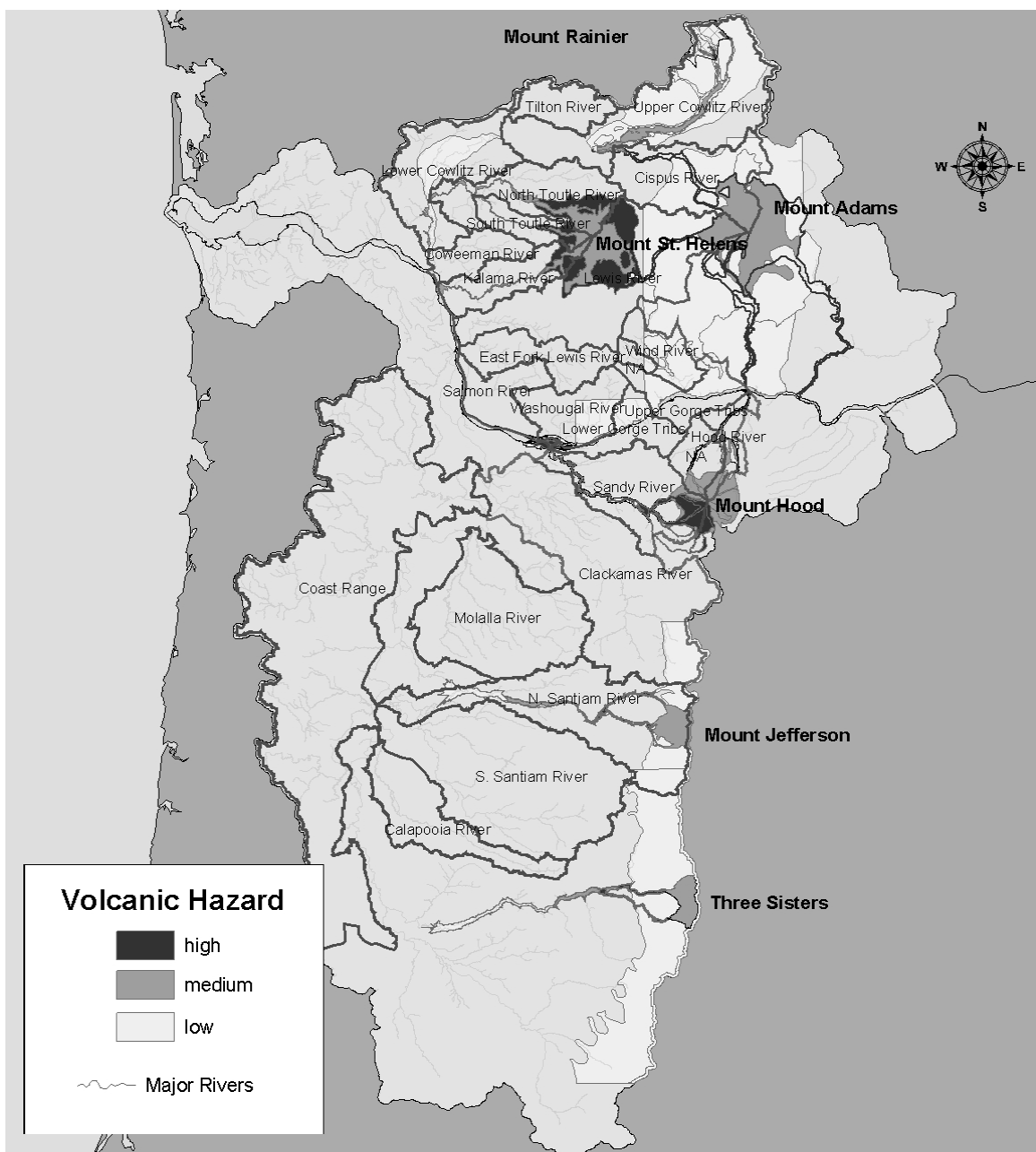


Figure K.3 Winter steelhead—Lower Columbia and Willamette River ESUs volcanic hazards.

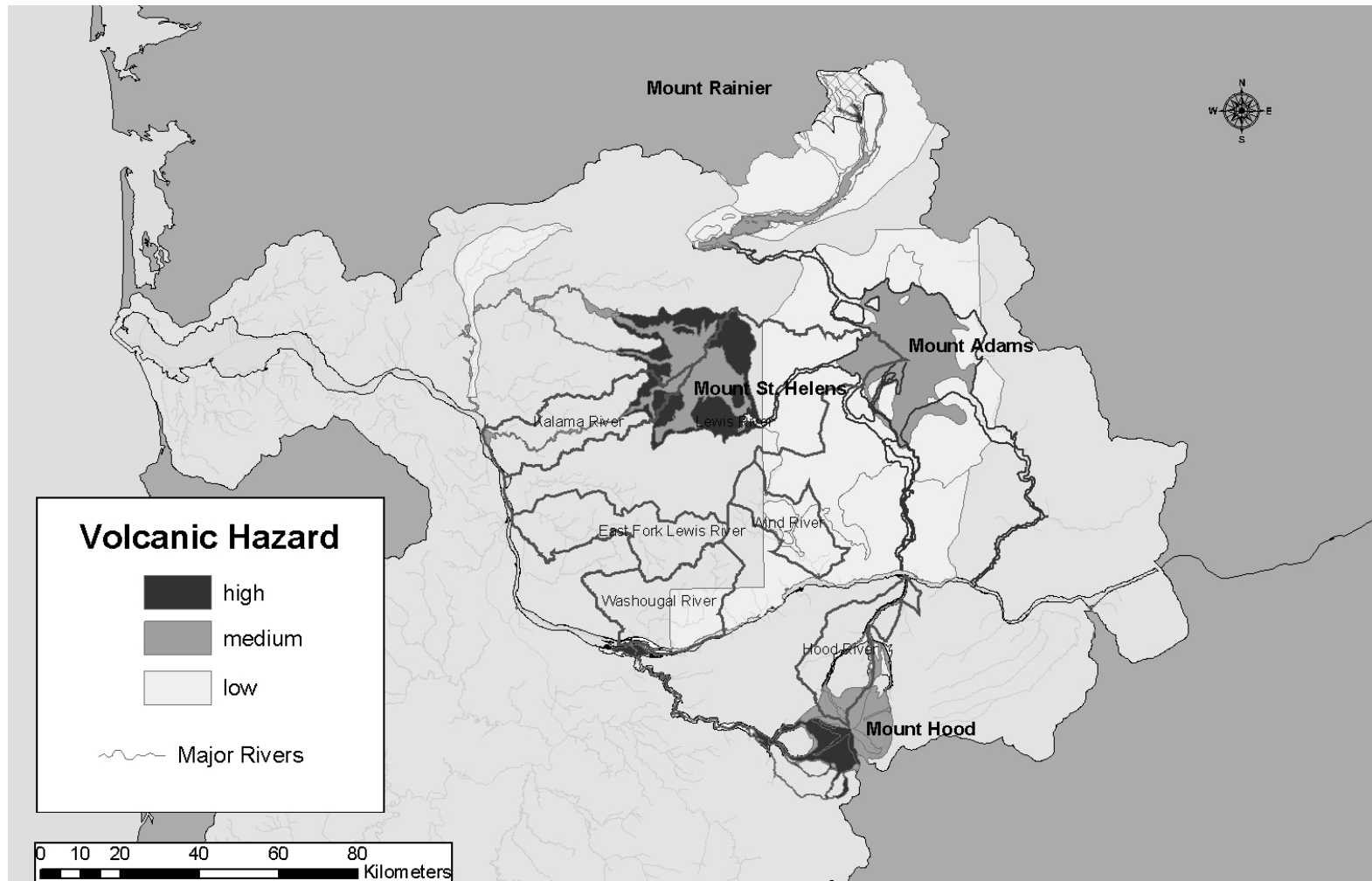
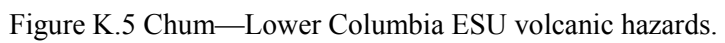


Figure K.4 Summer steelhead—Lower Columbia ESU volcanic hazards.



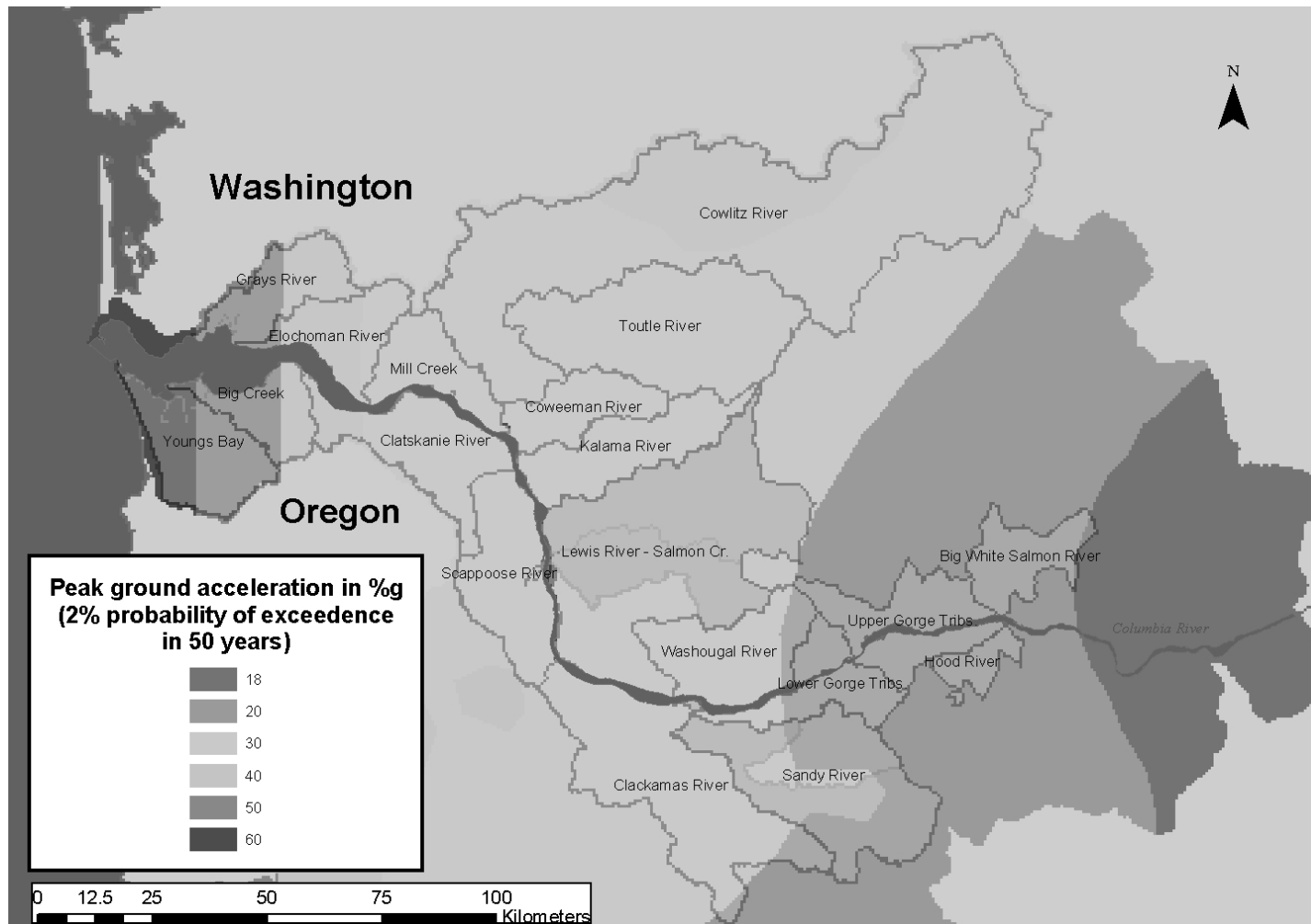


Figure K.6 Earthquake probabilities for fall chinook populations.

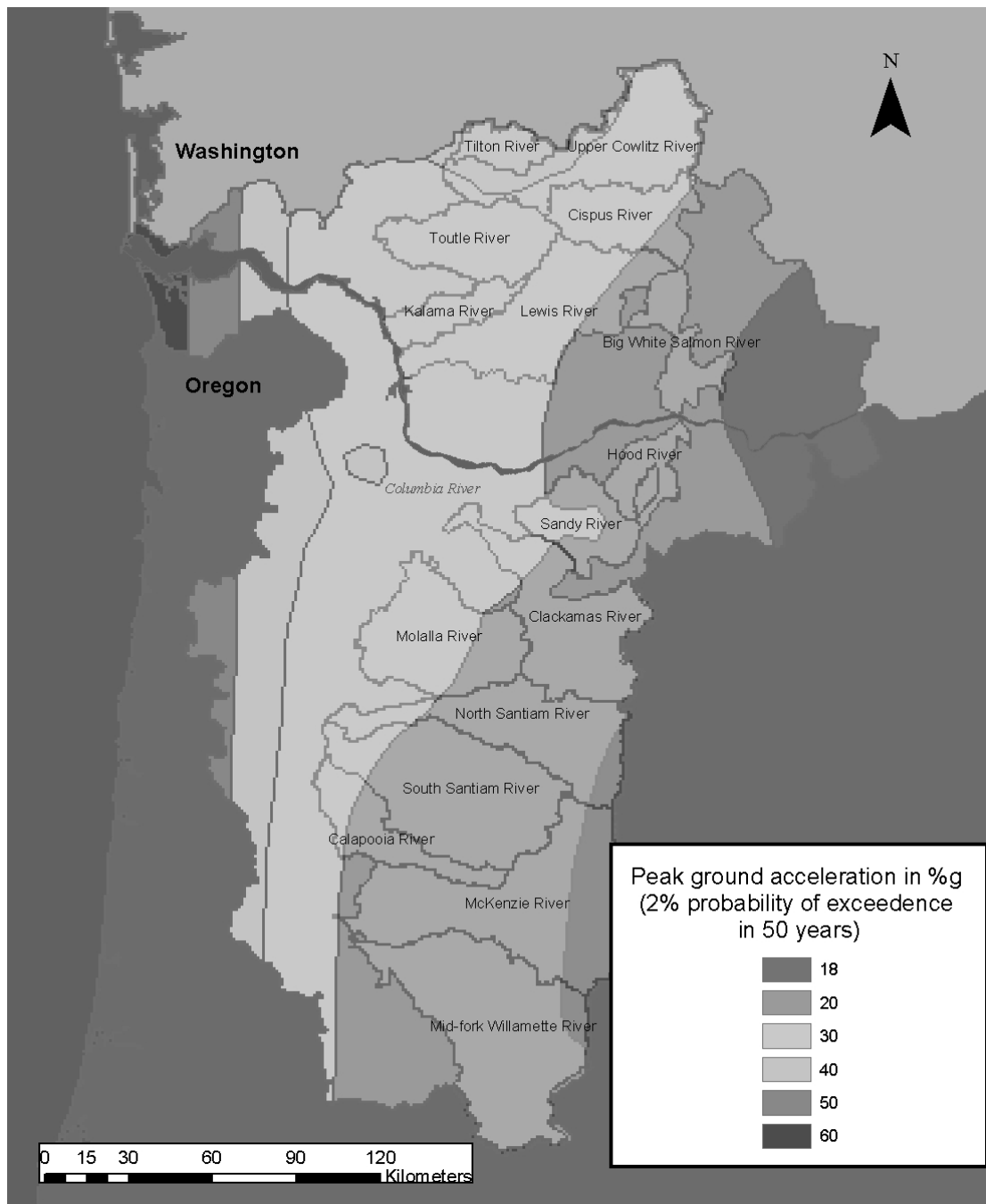


Figure K.7 Earthquake probabilities for spring chinook populations.

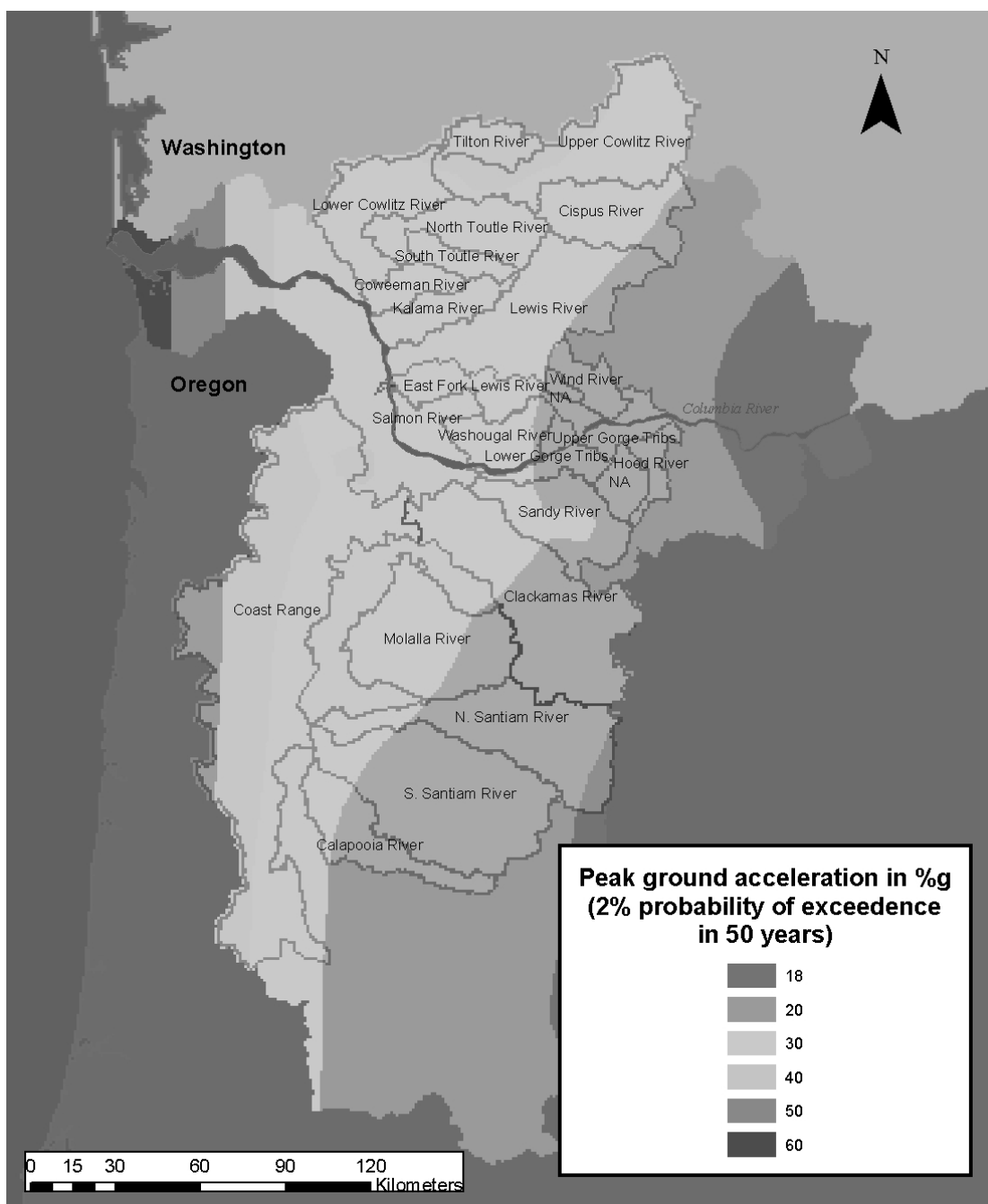


Figure K.8 Earthquake probabilities for winter steelhead populations.

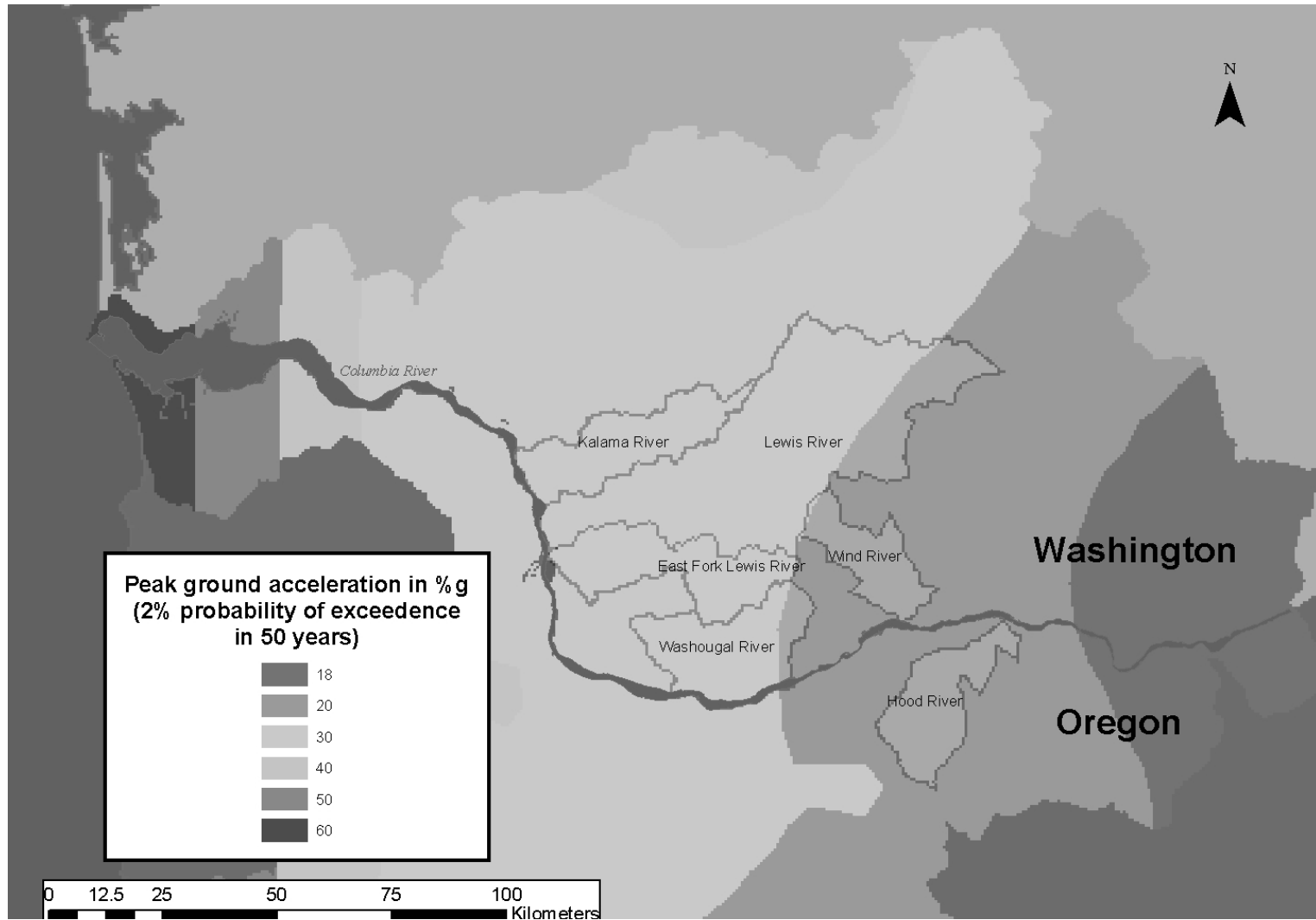


Figure K.9 Earthquake probabilities for summer steelhead populations.

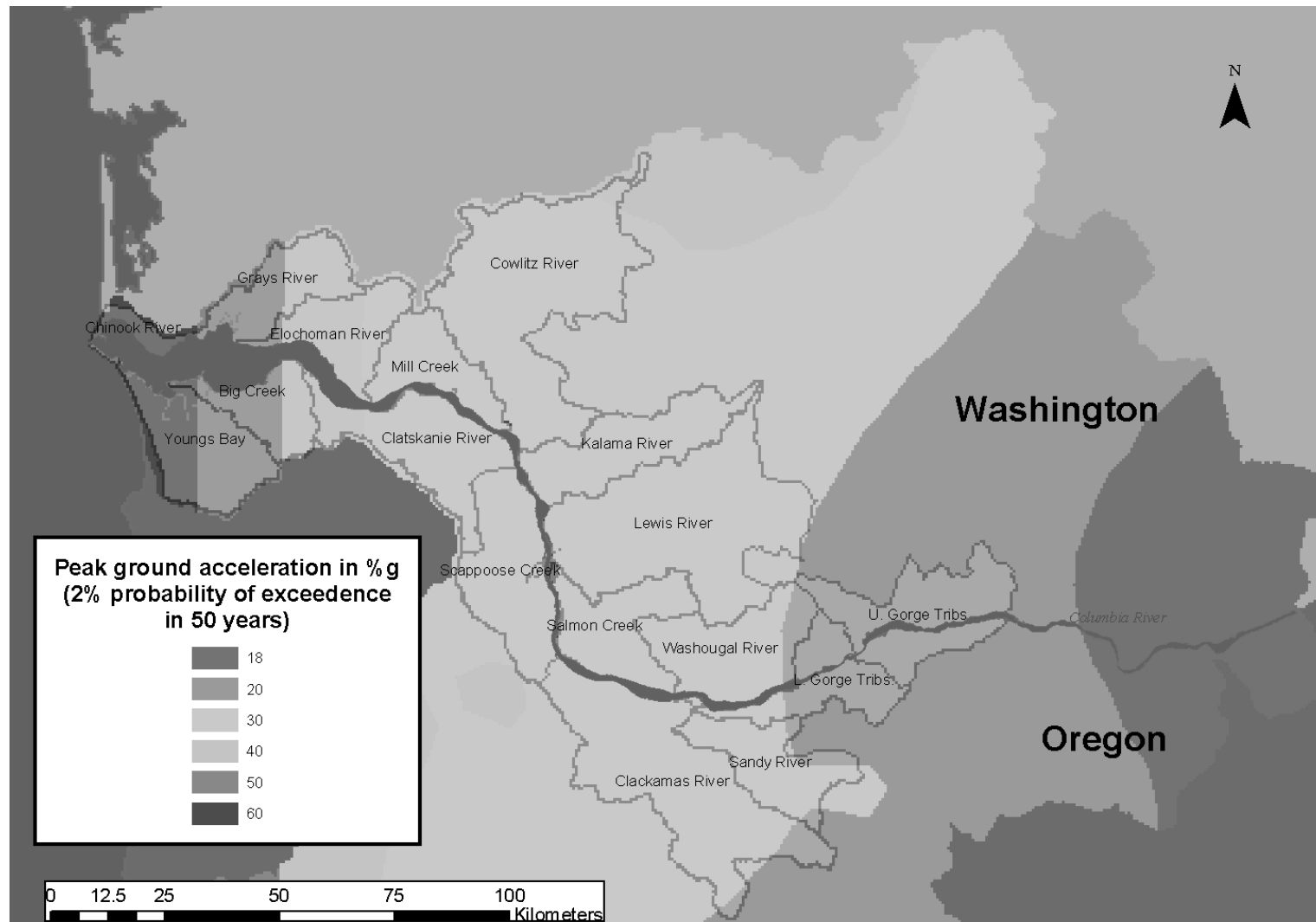


Figure K.10 Earthquake probabilities for chum populations.

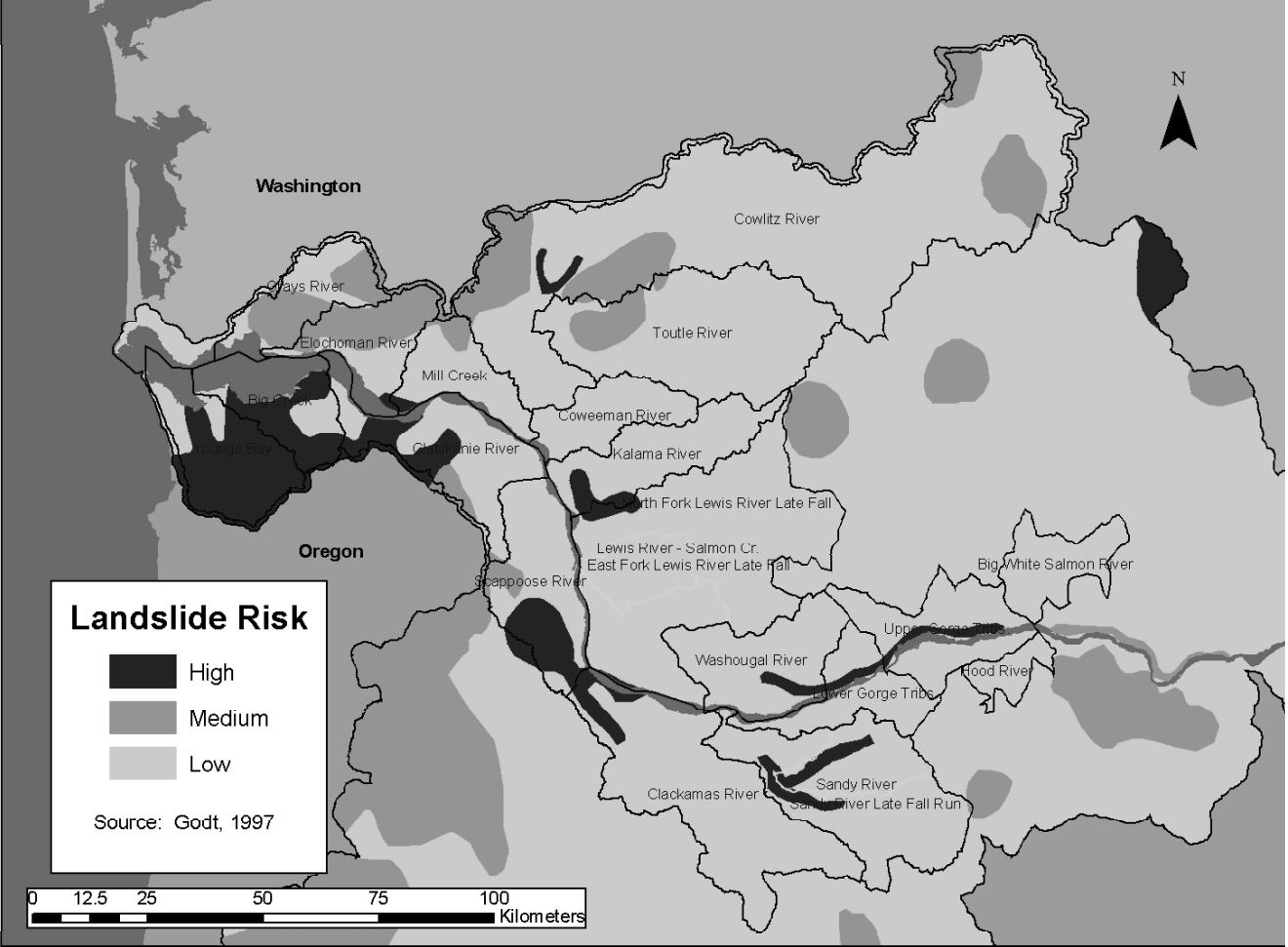


Figure K.11 Landslide risk to fall chinook populations.

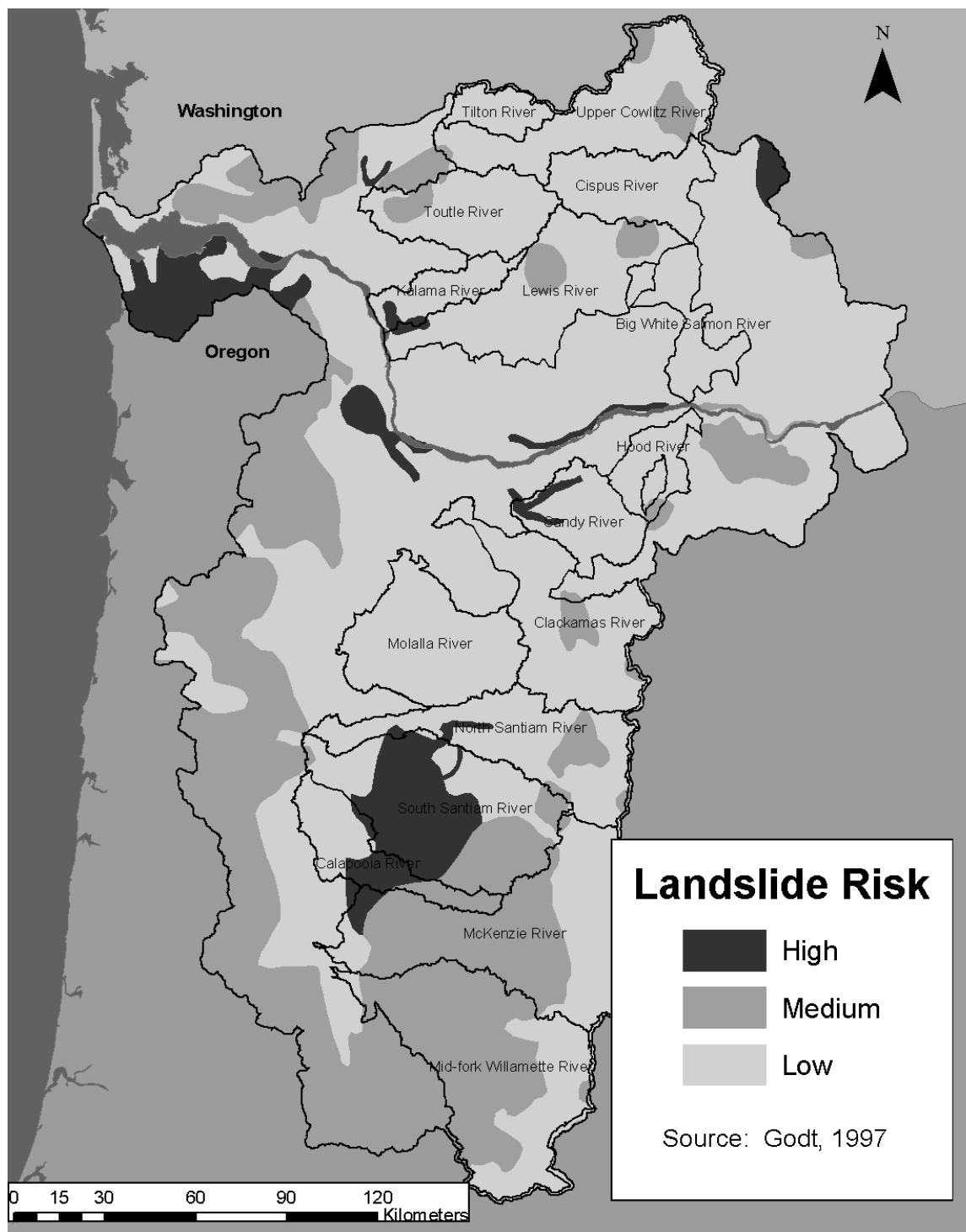


Figure K.12 Landslide risk to spring chinook populations.

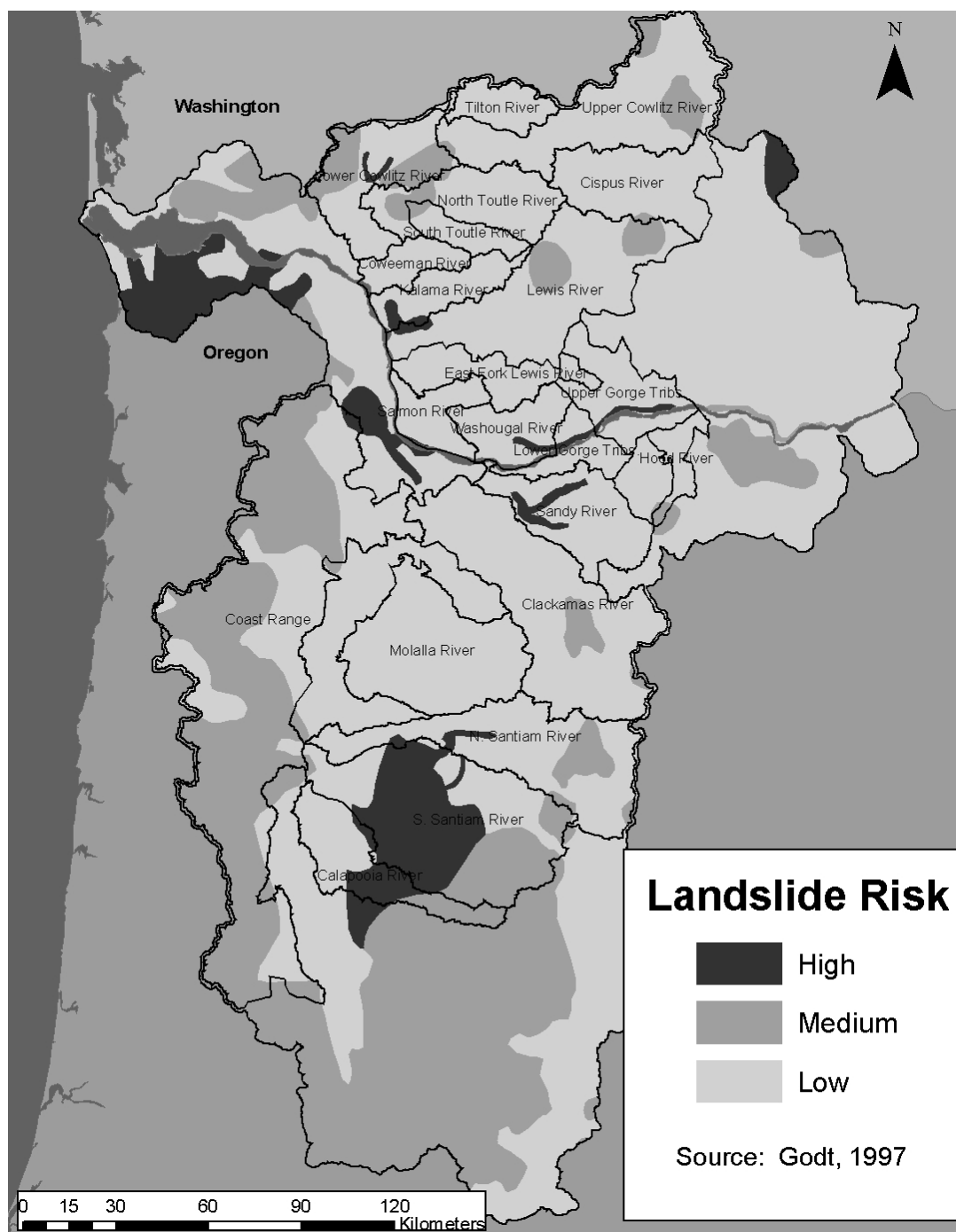


Figure K.13 Landslide risk to winter steelhead populations.

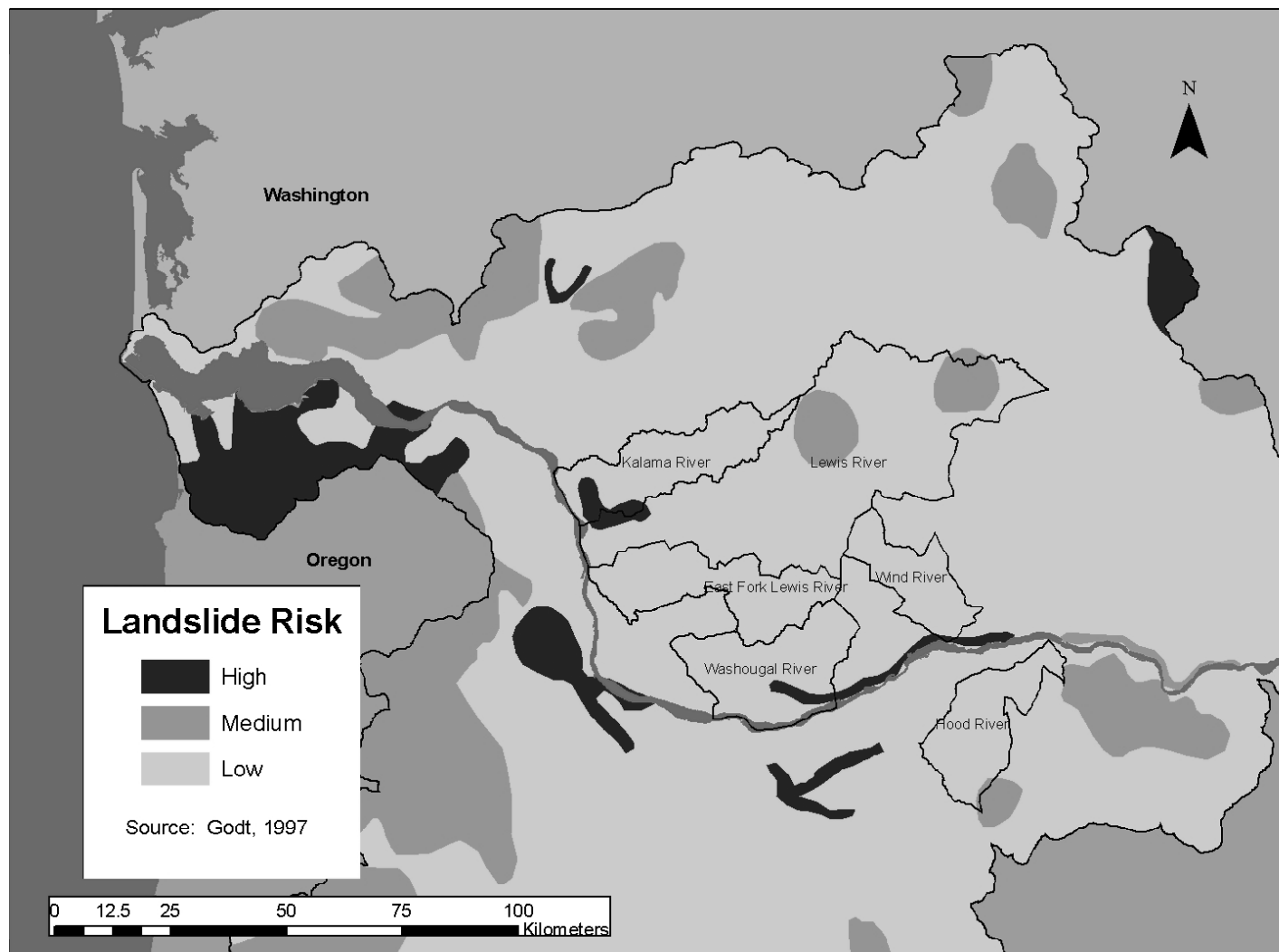


Figure K.14 Landslide risk to summer steelhead populations.

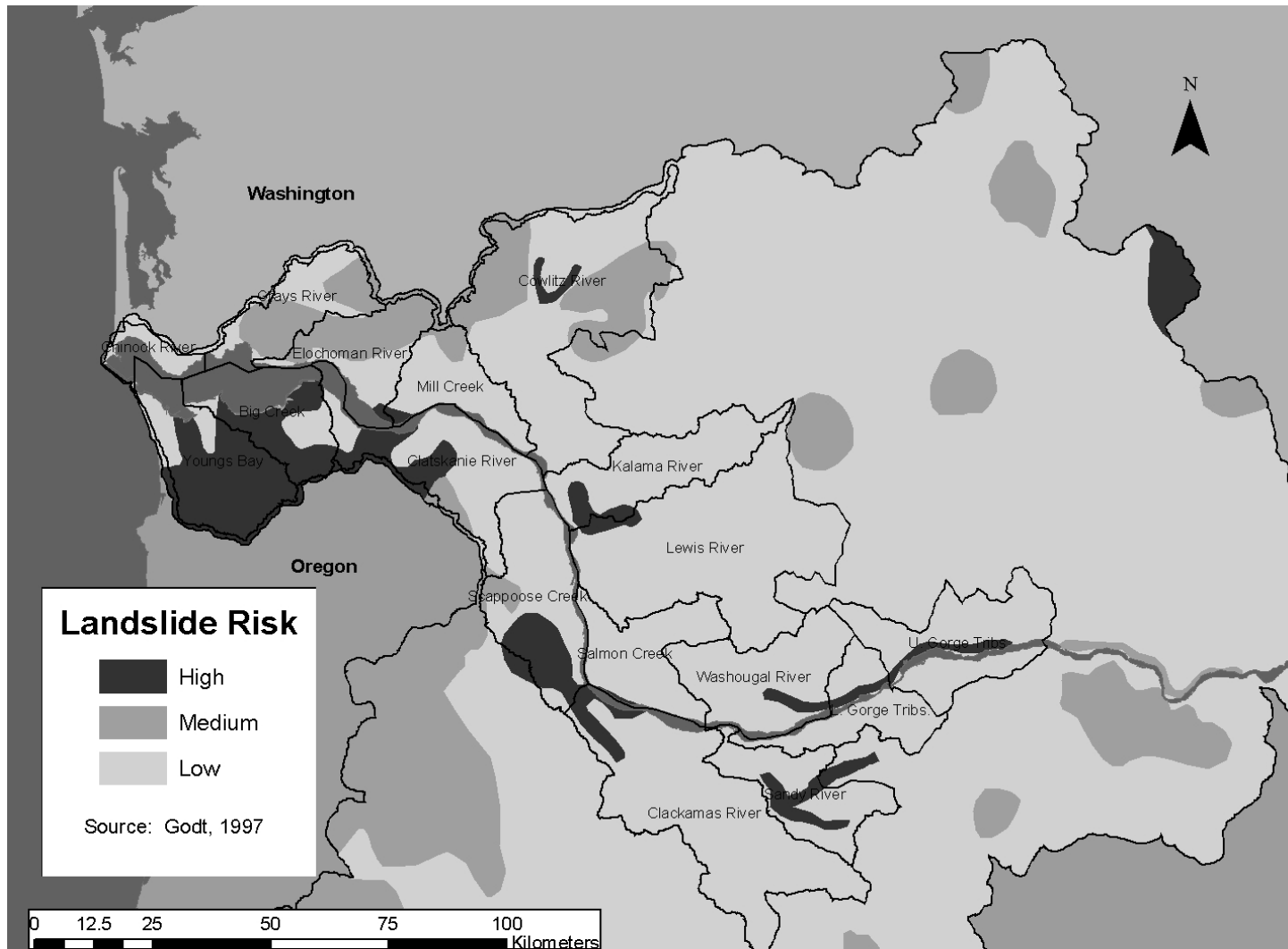


Figure K.15 Landslide risk to chum populations.

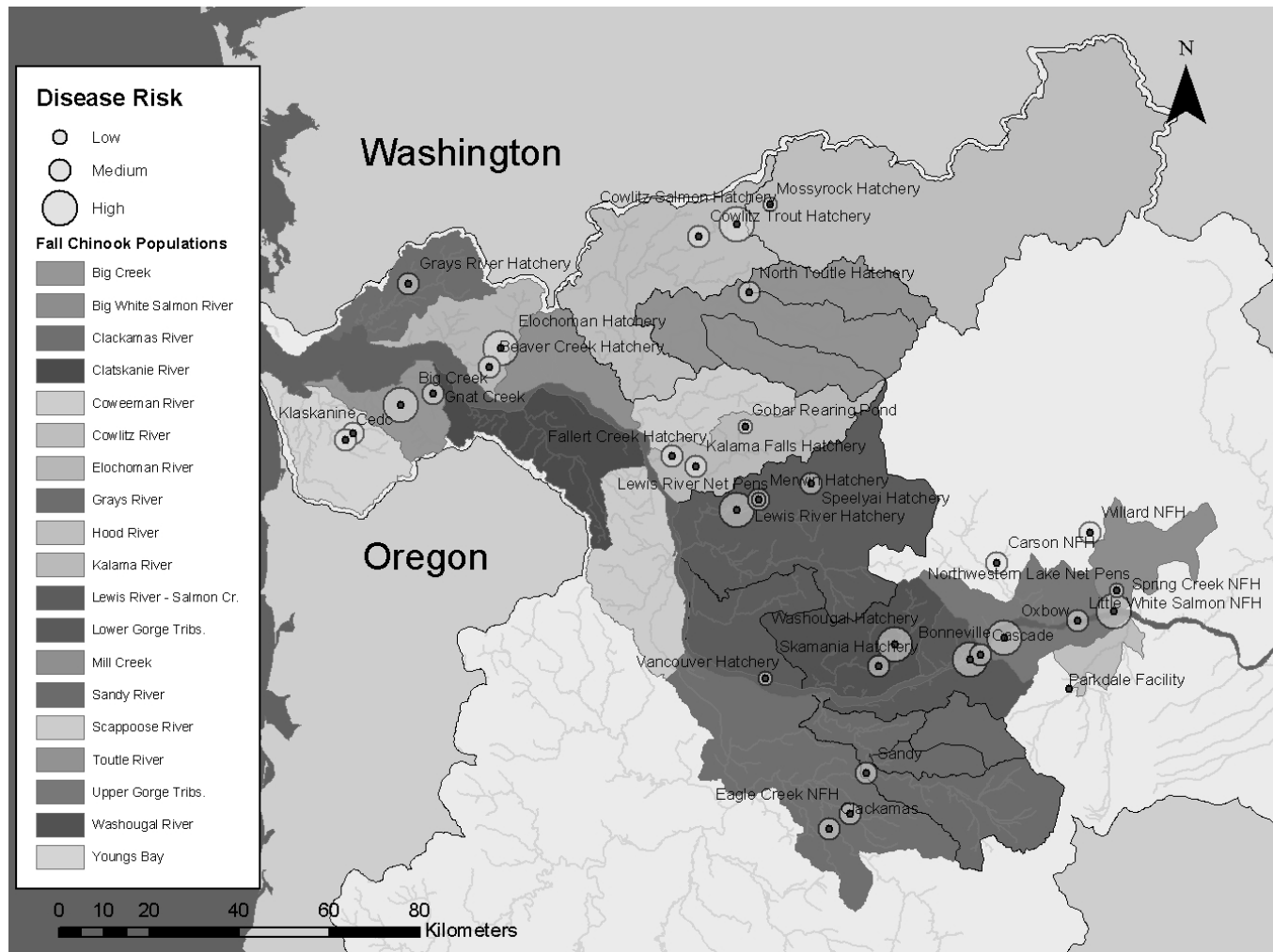


Figure K.16 Disease risk from hatchery releases to fall chinook.

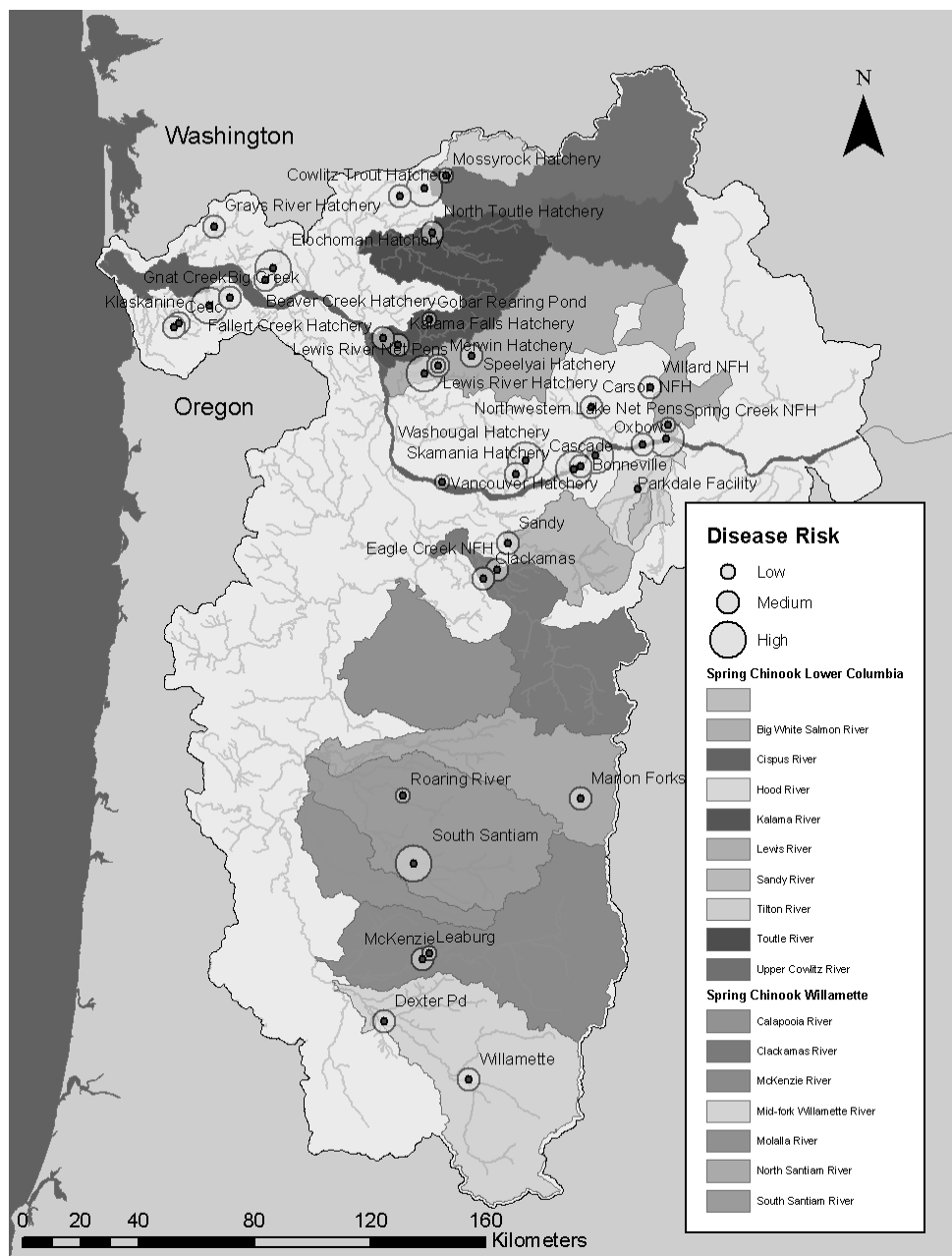


Figure K.17 Disease risk from hatchery releases to spring chinook.

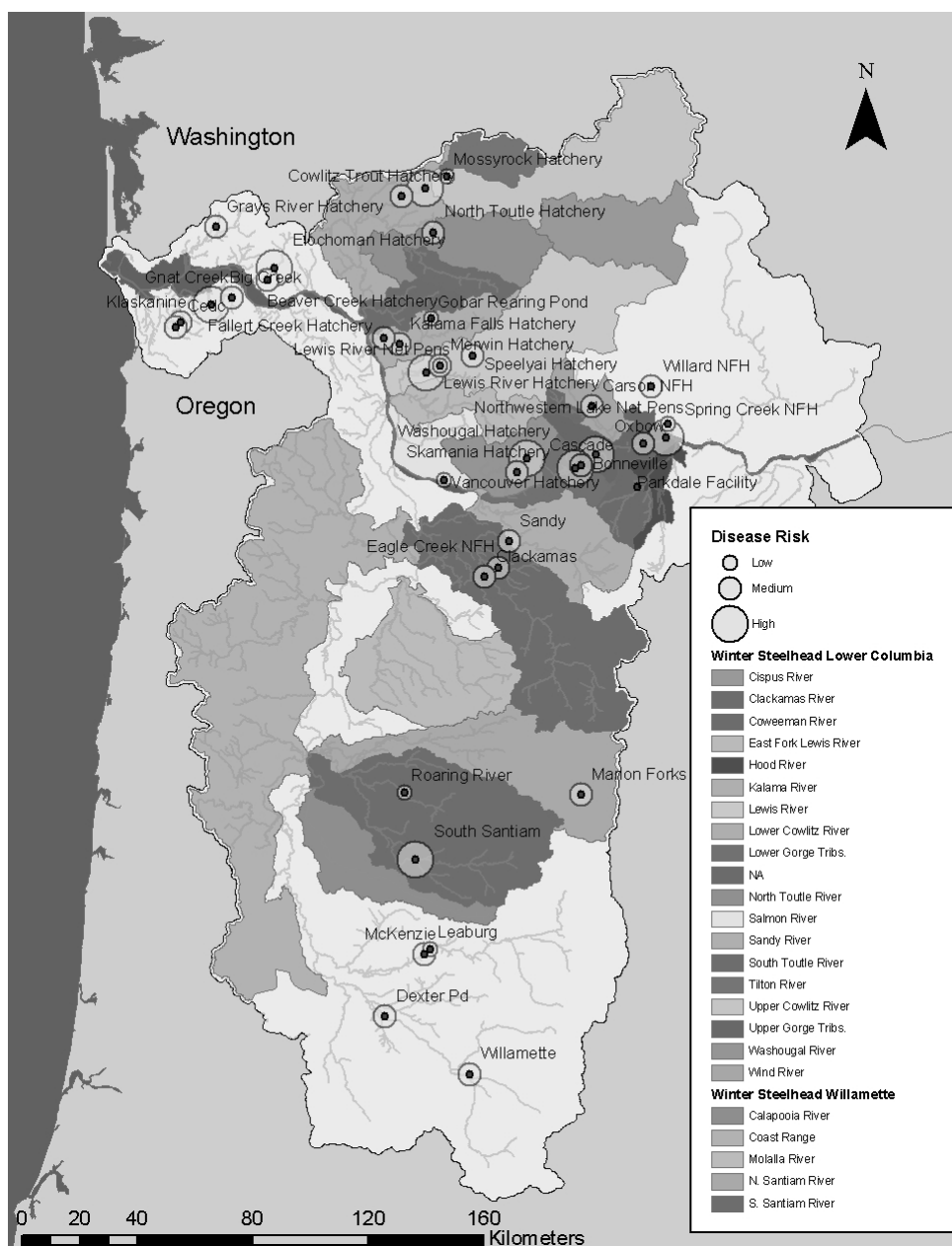


Figure K.18 Disease risk from hatchery releases to winter steelhead.

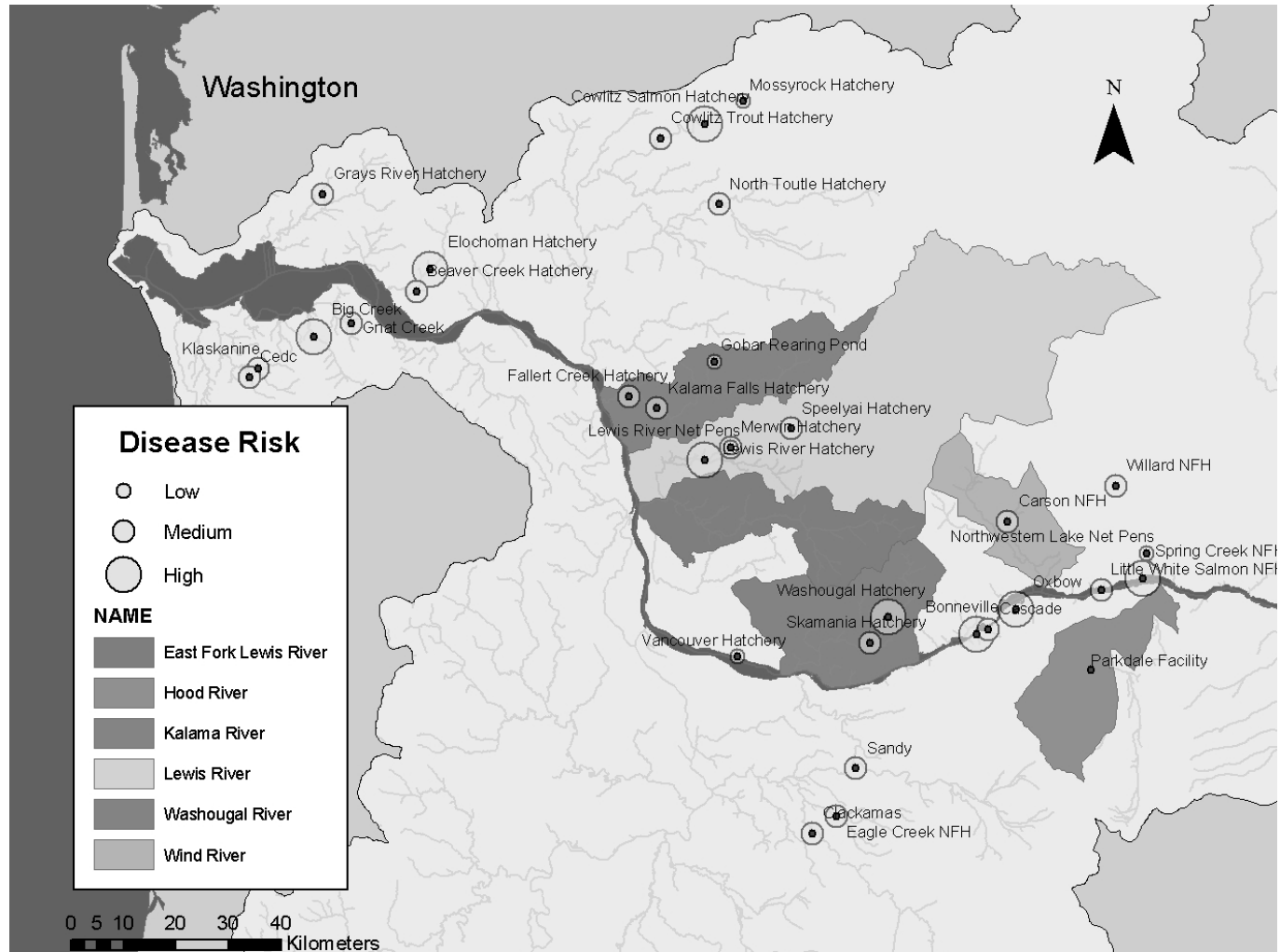


Figure K.19 Disease risk from hatchery releases to summer steelhead.

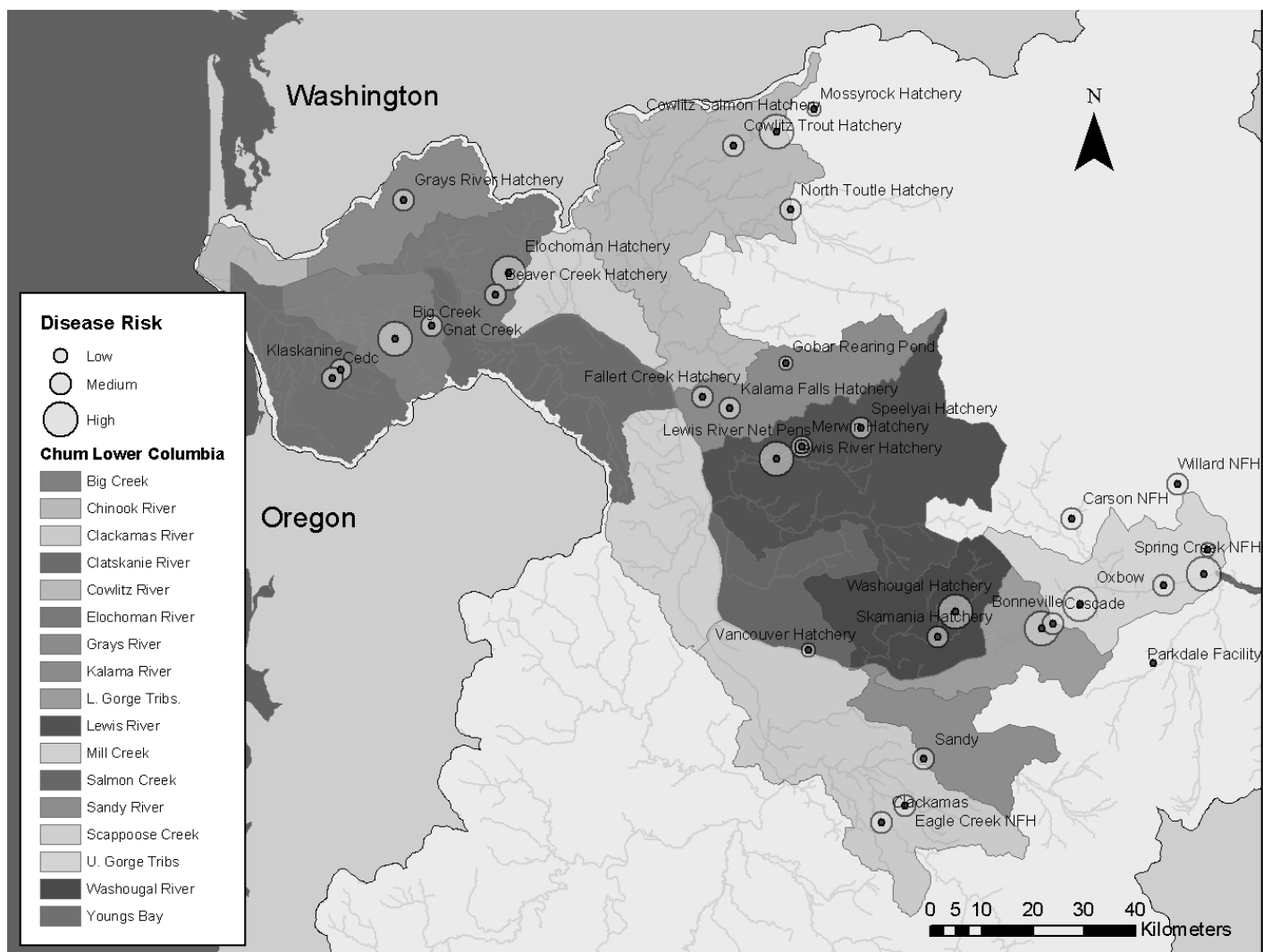


Figure K.20 Disease risk from hatchery releases to chum.

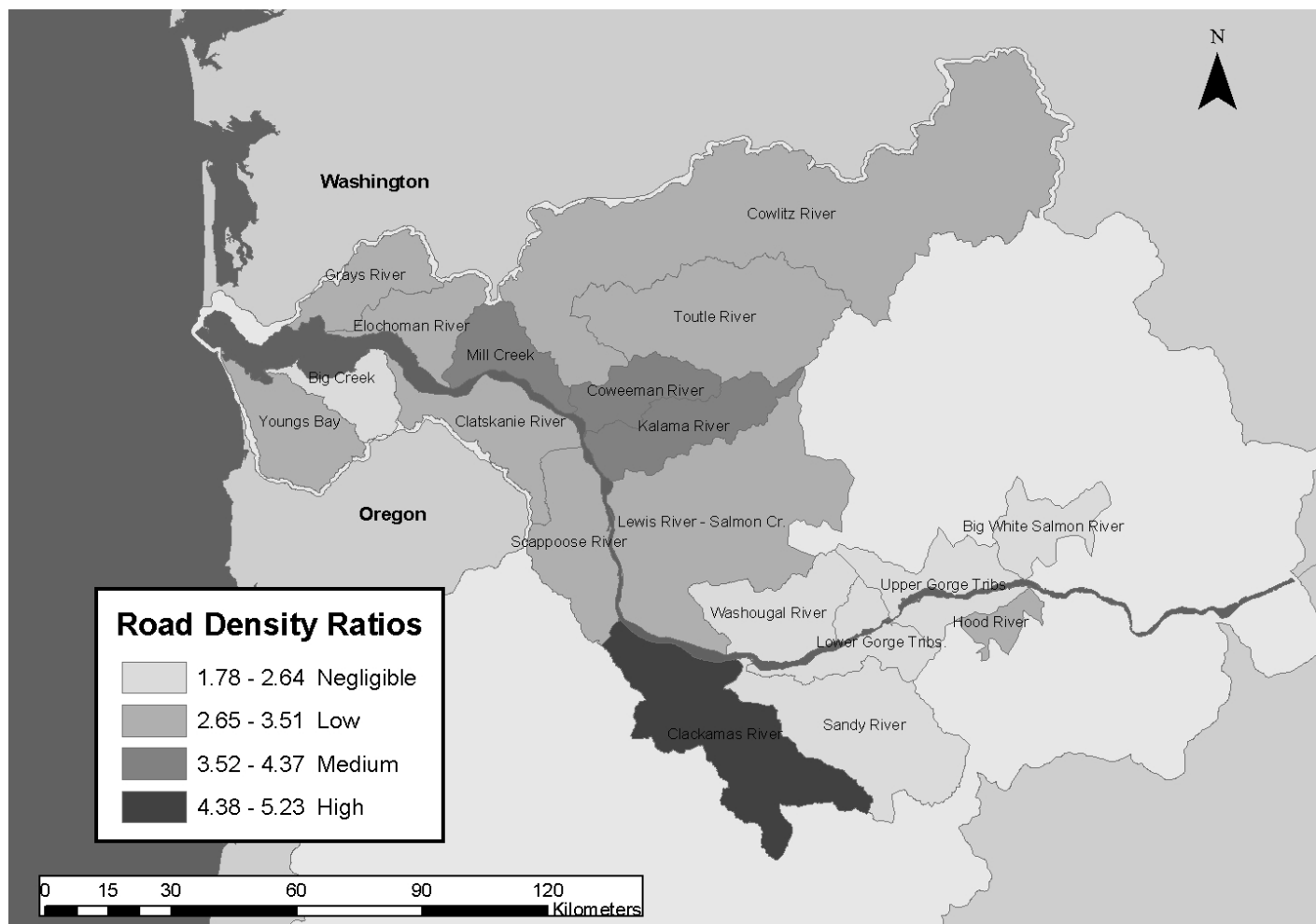


Figure K.21 Relative road density ratios by fall chinook populations.

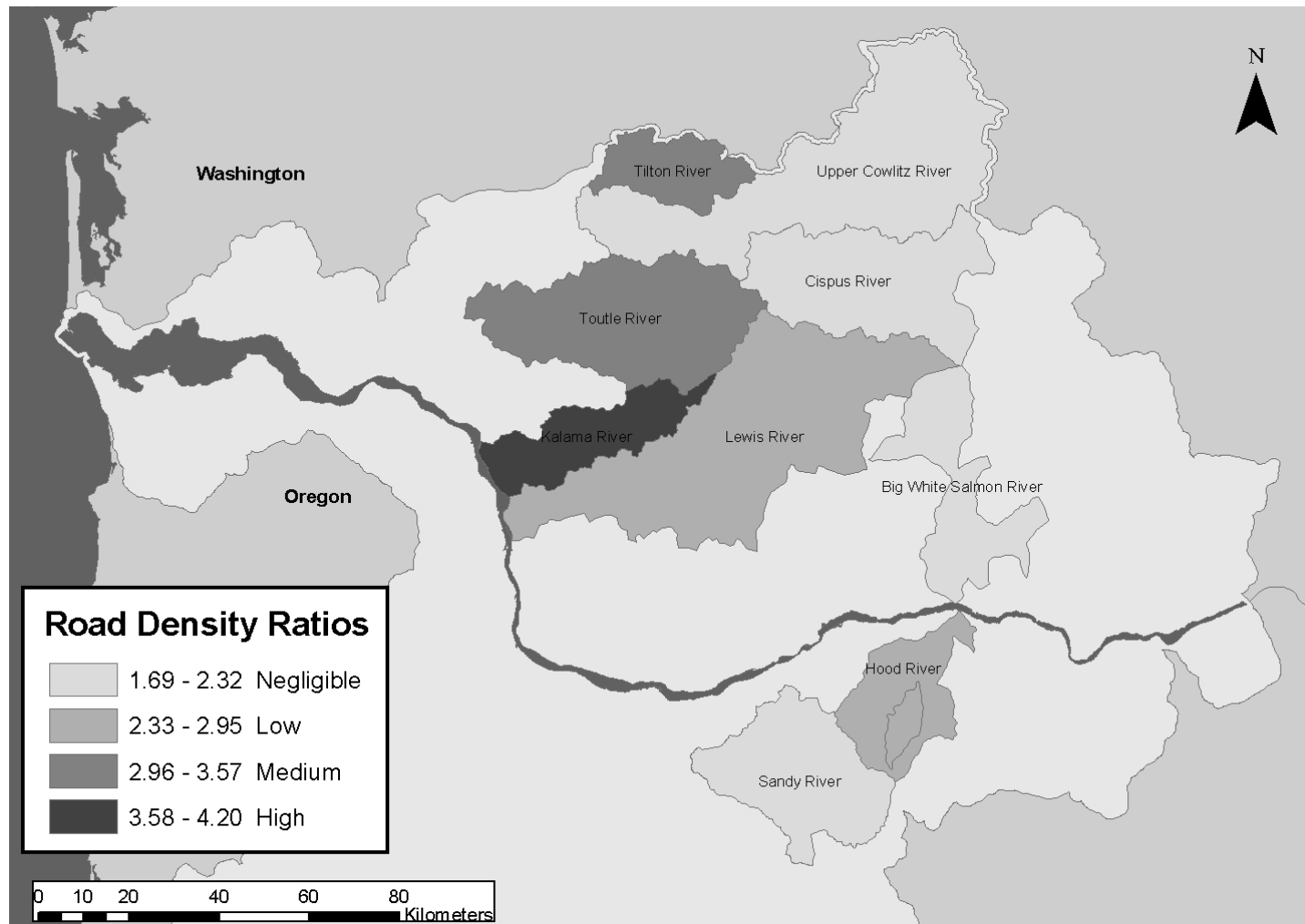


Figure K.22 Relative road density ratio by Columbia River spring chinook populations.

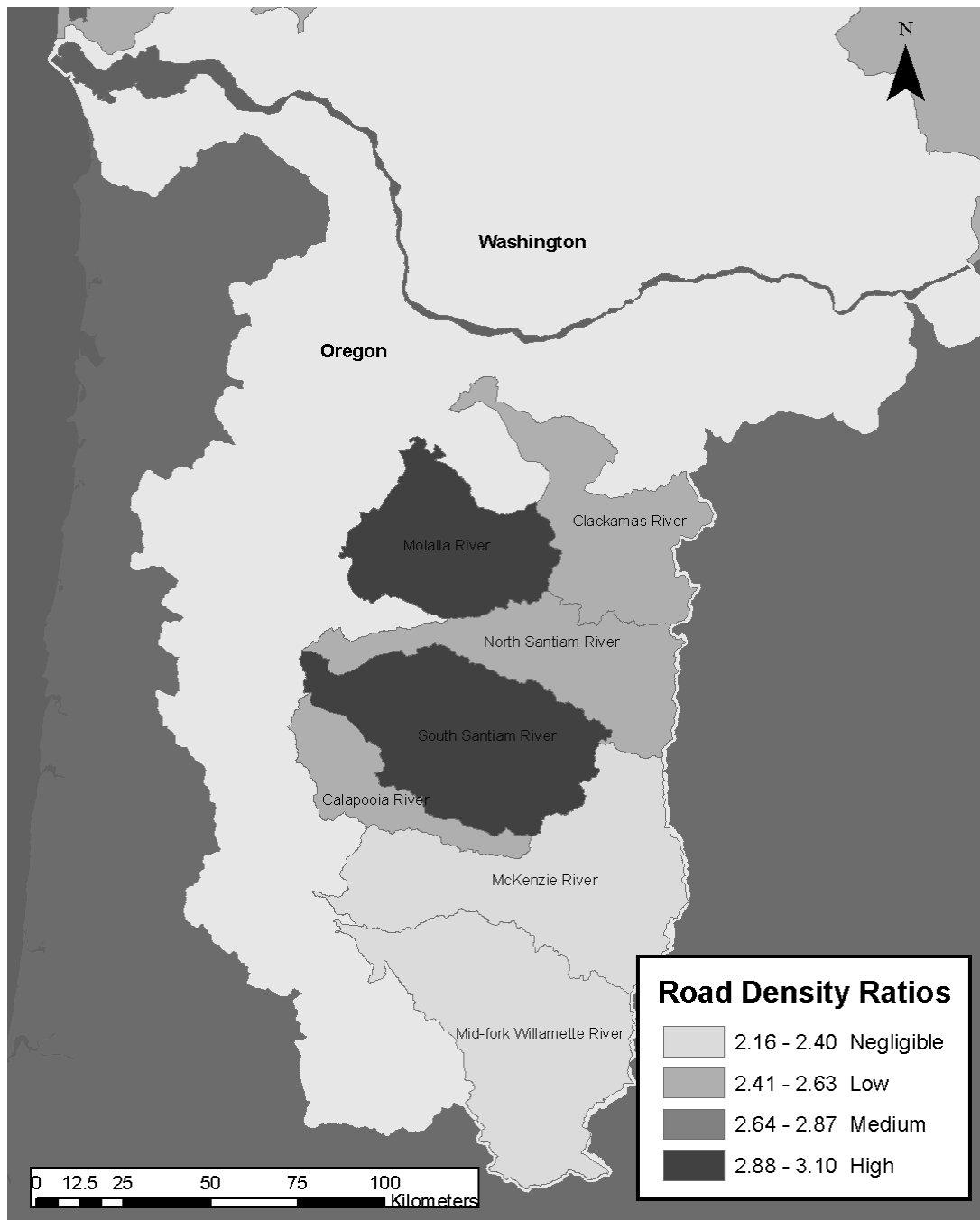


Figure K.23 Relative road density ratio by Willamette spring chinook populations.

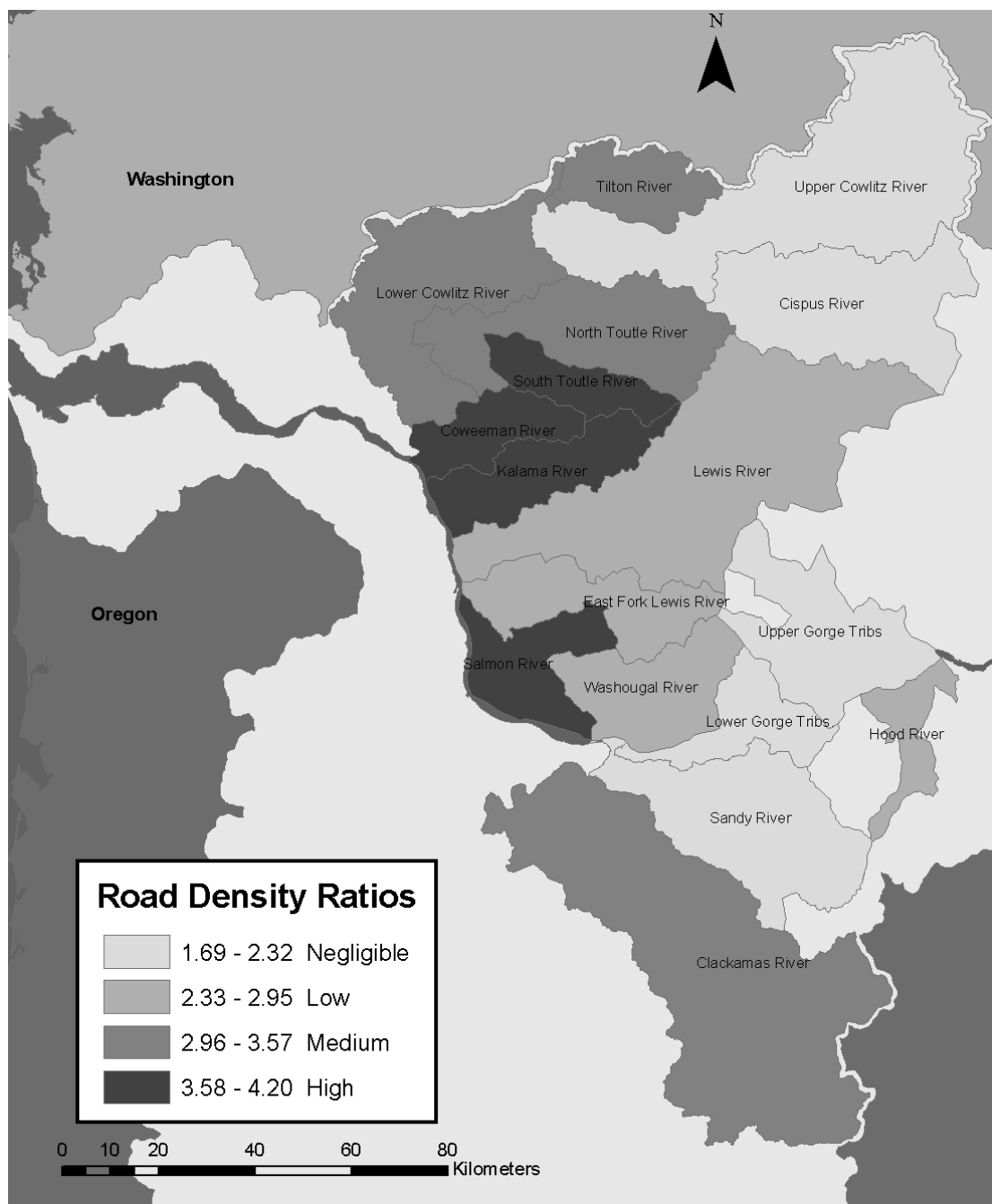


Figure K.24 Relative road density ratios by Lower Columbia winter steelhead populations.

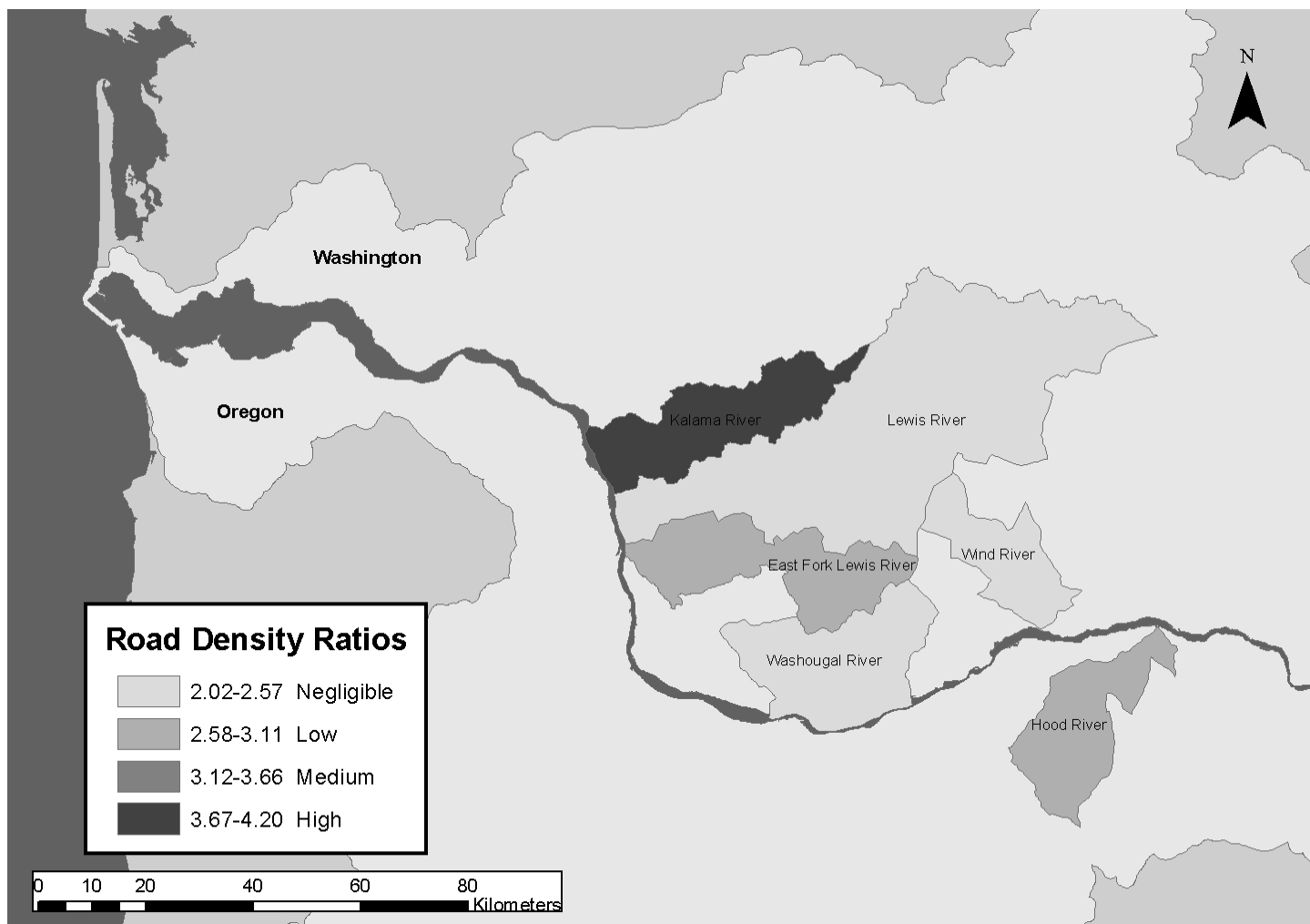


Figure K.25 Relative road density ratios by summer steelhead populations.

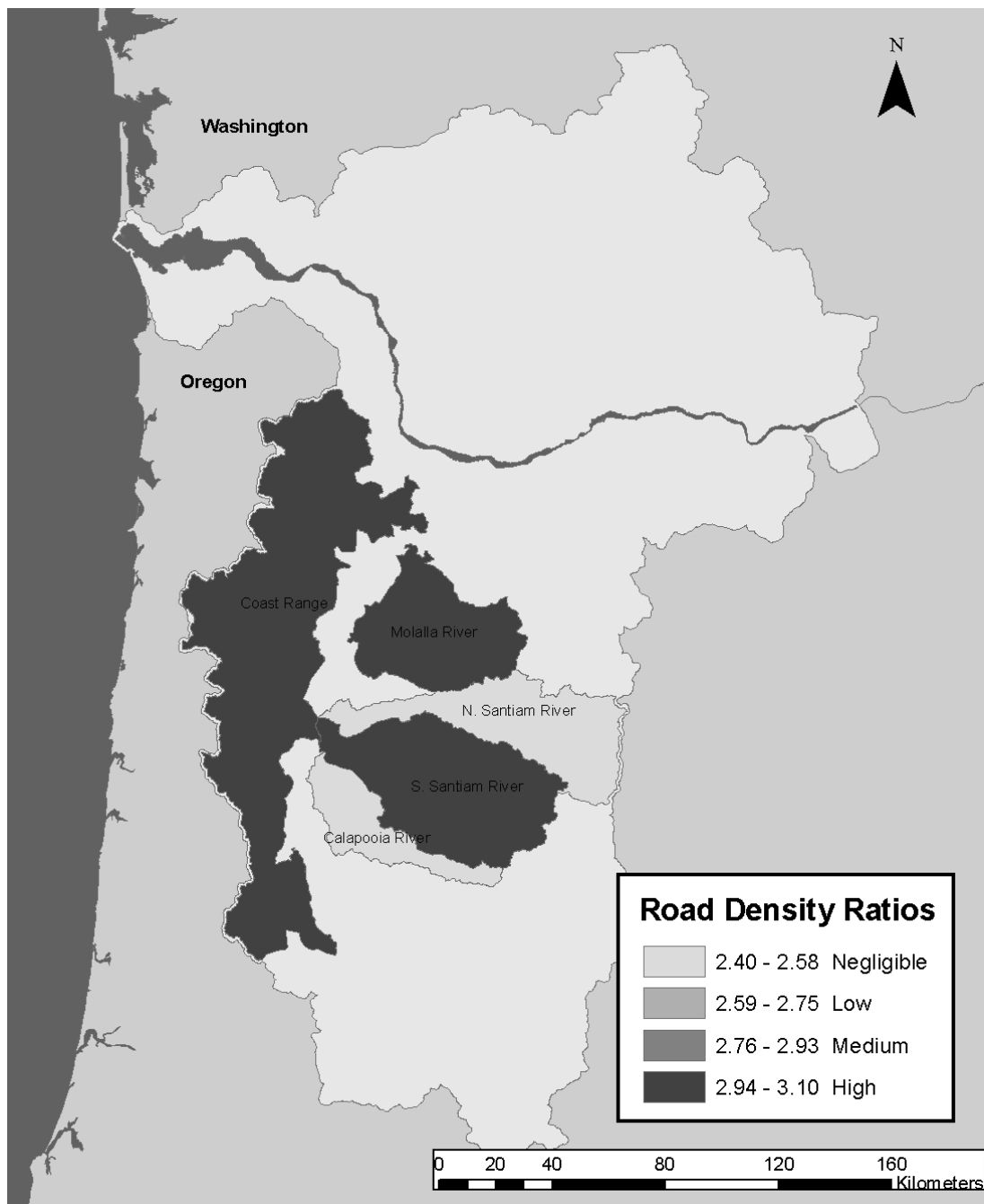


Figure K.26 Relative road density ratios by Willamette winter steelhead populations.

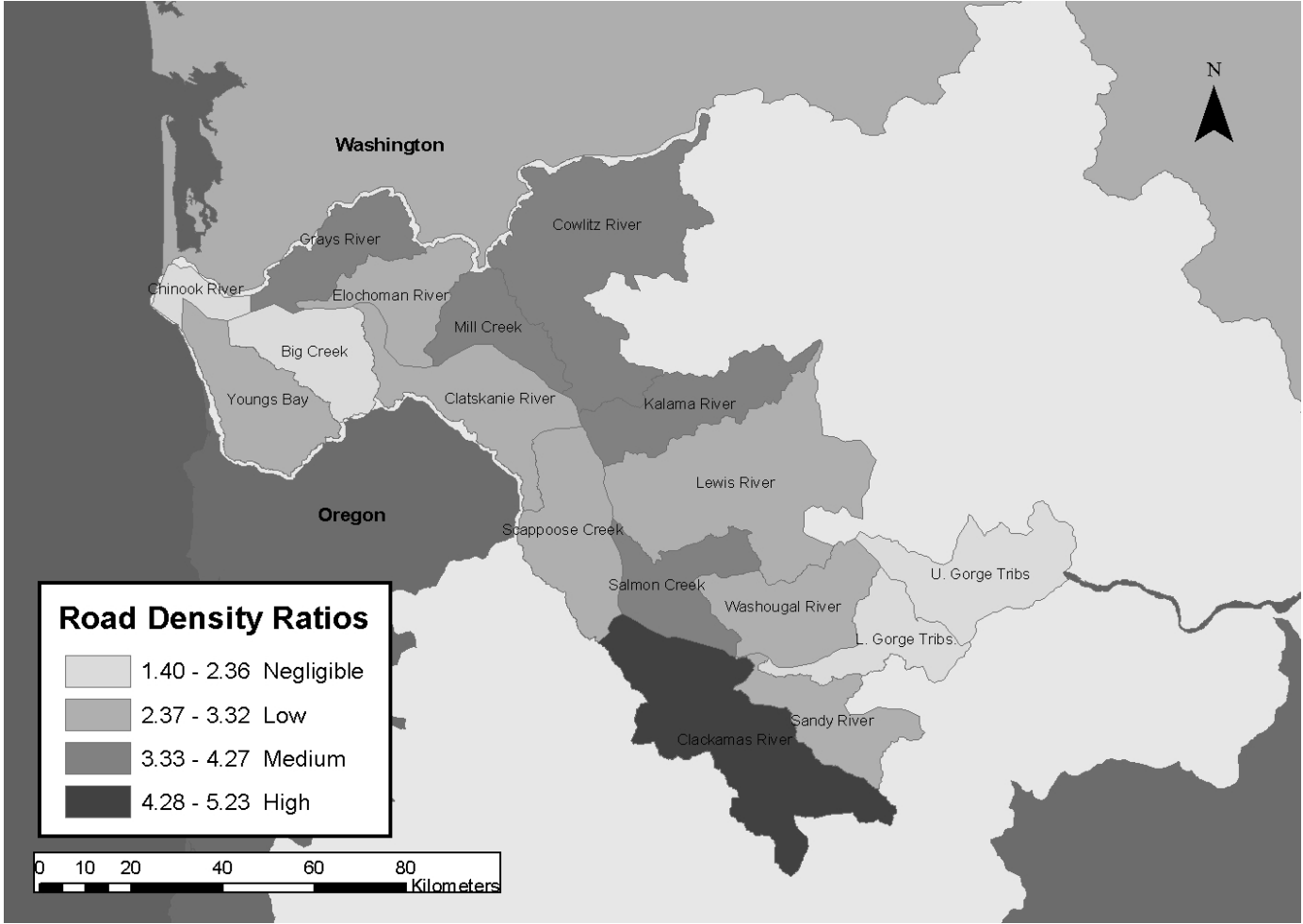


Figure K.27 Relative road density ratios by chum population.

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APPENDIX L

MAXIMUM TEMPERATURE: UPPER OPTIMAL TEMPERATURE LIMITS FOR SALMONIDS IN THE WILLAMETTE AND LOWER COLUMBIA RIVERS

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Introduction

The Willamette/Lower Columbia River Technical Recovery Team (WLC-TRT) is responsible for recommending delisting criteria. As these criteria need to address all the factors for species listing, they must make reference to habitat.

Adequate water quality is important to all salmonids at all life-history stages. The distributions of native salmonid fish in the Pacific Northwest are strongly tied to temperature conditions in their habitats. Because water temperature affects the health of individual fish, it also affects entire populations and species assemblages. Temperature may directly affect salmonids in obvious ways, or indirectly through interaction with other important variables (Dunham et al. 2001). For example:

- Given sufficient magnitude and time, high temperatures can cause weight loss, disease, competitive displacement by species better adapted to the prevailing temperature, or death (Sullivan et al. 2000).
- When fish are stressed by any one process, they are less able to deal with other stressors. Salmonids already stressed by high water temperature will be less able to deal with a second stressor (e.g., toxic pollutant, pathogen). Warmer temperatures often increase the infection rate or virulence of fish pathogens and lessen the ability of a fish to withstand disease (Materna 2001).

Human Influence on Thermal Regimes

In many streams that once were inhabited by large salmon runs, temperature regimes are now inhospitable. An important factor in the recovery of salmonid populations is the restoration of temperature regimes (Poole et al. 2001a, b).

Human activities can affect thermal regimes by simplifying the physical structure of aquatic systems, thereby eliminating natural thermal buffers and insulators (Poole and Berman 2001). Clearing and developing land, dredging or straightening streams, grazing and other land-

use activities influence temperature regimes by altering factors external to the stream, structures within the stream, and the amount of water flowing in the stream (Poole et al. 2001a, b). These activities often directly or indirectly simplify the structure of stream channels or riparian zones, as has occurred in the lower Willamette River (Sedell and Froggatt 1984). This type of channel simplification can potentially increase temporal variability and decrease fine-scale spatial variability in stream temperature, both of which may have negative consequences for salmonids (Poole et al. 2001b, Poole and Berman 2001). Removing riparian vegetation in small streams, where shading is important, can increase daily variation in stream temperature (Beschta 1997). For streams where groundwater buffers temperature, change in groundwater temperature or flow dynamics can alter the seasonal availability of cold water, including increased seasonal variation in water temperature. Small-scale thermal refugia can provide important habitat for salmonids during periods of warmer water temperatures (Ebersol 2002), but even slight changes in temperature extremes, or mean temperature, can result in refugium, and therefore salmonid, loss. Changes in the timing of maximum and minimum temperatures can occur with or without associated changes in the actual values of maximum, minimum, or mean stream temperatures, and these too can threaten salmonids because of their sensitivity to temperature at many life stages. Water temperature is an indicator of habitat quality, which is an integrator of what is happening in a watershed.

Thermal refugia are important in maintaining salmonid populations because when daily variation in stream temperature is high, salmonids may be exposed to stressful or lethal temperatures for part of the day. Thermal refugia provide protection for salmonids when temperatures are extreme (Ebersol 2002). At peak summertime temperatures, only a small percentage of habitat in some streams may be cool enough. Loss of riparian vegetation, the elimination of large beaver populations, removal of large woody debris, channel simplification, reduced groundwater discharge due to changes in upland vegetation, water withdrawals, and other human activities cause the loss of the fine-scale spatial distribution of appropriate thermal habitats upon which salmon rely (Poole et al. 2001b). This can cause fish to migrate greater distances to find appropriate habitats or not find them at all.

In the same way, seasonal variation in temperature can create thermal barriers to salmonid immigration and emigration. Anthropogenic activities can increase the coarse-scale temporal variation of streams, exposing salmonids to extremes beyond the normal range of variation and resulting in habitat fragmentation and elimination of the large, well-connected tracts of high-quality thermal habitat. This habitat fragmentation has been shown to degrade both population structure and persistence (Poole et al. 2001b).

In a recent draft document entitled Draft EPA Region 10 Guidance for State and Tribal Water Quality Standards,¹ the U.S. Environmental Protection Agency (EPA) recommended a four-part approach for state and tribal temperature standards to support native salmonids. This approach includes the adoption of:

1. thermal, potential numeric criteria for bodies of water, which are estimations (generally on the subbasin scale) of the thermal potential of bodies of water based on an average meteorological year, with adjustments for other climatic conditions;
2. interim, species-life-stage numeric criteria as a bridge until a newer approach to temperature criteria is developed;

¹ EPA. 2001. available at <http://yosemite.epa.gov/R10/WATER.NSF/1507773cf7ca99a7882569ed007349b5/ce95a3704aeb5715882568c400784499?OpenDocument>.

3. temperature management plan provisions; and
4. provisions to protect existing cold-water areas.

The EPA criteria are based on salmonid guilds. These guilds are described by the EPA as groups of species sharing similar life strategies, with similar temperature and habitat needs and limitations. In the Pacific Northwest, the cold-water guild includes the five Pacific salmon, anadromous steelhead trout, and coastal cutthroat and rainbow trout. This document discusses the species-specific data that exist, and then uses them as background criteria to support the EPA's recommended guild approach. The EPA draft criteria are being considered for application to potential salmonid habitat, as distinct from present or historical salmonid habitat. Potential habitat has not been explicitly identified by the EPA, but consists of those areas salmonids might inhabit without the dismantling of major barriers to passage. Whether the use of the potential habitat concept comes to closely match the spatial distribution and population number requirements eventually adopted by the National Marine Fisheries Service (NMFS) for delisting threatened or endangered salmonids, the guild approach associated with it can be evaluated separately in terms of maximum temperature criteria for salmonid delisting.

This paper will summarize the large body of information about thermal effects on salmonids, specific to life stage and species, and use that information to propose draft water temperature criteria. Several groups that recently produced white papers on the topic are the Pacific Northwest Environmental Indicator Work Group (PNWEIWG), the Sustainable Ecosystems Institute (SEI), the EPA Water Temperature Criteria Technical Workgroup, the Columbia River Inter-Tribal Fish Commission, and the Washington State Department of Ecology. The participating agencies in the PNWEIWG are the British Columbia Ministry of Environment, Lands, and Parks; the Alaska Department of Environmental Conservation; the Idaho Division of Environmental Quality; the Washington State Department of Ecology (WDOE); Environment Canada; and the U.S. EPA (Region 10). In 1997, directors of these agencies asked the PNWEIWG to pilot development of regional indicators associated with risks to salmonid stocks. Indicators were required to: (1) have data available, (2) be integral to measuring the performance of salmon issues for PNWEIWG agencies, and (3) be able to be reported cost-effectively in a monitoring program.

Martin Environmental, Parametrix Inc., and Weyerhaeuser Company participated in the SEI, which was funded by the Oregon Forest Industries Council, Washington Forest Protection Association, and Weyerhaeuser Company. The SEI developed a risk-based approach to analyze summer temperature effects on juvenile salmon species in their December 2000 publication, *An Analysis of the Effects of Temperature on Salmonids of the Pacific Northwest with Implications for Selecting Temperature Criteria* (Sullivan et al. 2000). This report reviewed the aspects of temperature affecting the rearing of salmonid species in the freshwater environment and discussed lethal (acute) as well as sublethal (chronic) effects. The main focus of the report was on temperatures affecting growth and mortality (Sullivan et al. 2000).

The EPA established the Water Temperature Criteria Technical Workgroup to assist in developing temperature criteria guidance for EPA Region 10. The purpose of the EPA guidance is to help Pacific Northwest states and tribes adopt water temperature standards that (1) meet the biological requirements of native salmonid species (Pacific salmon, trout, and charr) for survival and recovery pursuant to the Endangered Species Act (ESA); (2) provide for the protection and propagation of salmonids under the Clean Water Act (CWA); and (3) meet the salmonid restoration goals of federal trust responsibilities with treaty tribes. The technical workgroup, a

panel of experts on salmonid biology and stream temperature, represented the following agencies: EPA, U.S. Forest Service, WDOE, NMFS, U.S. Fish and Wildlife Service, Columbia River Inter-Tribal Fisheries Commission, Idaho Department of Environmental Quality, Oregon Department of Environmental Quality, U.S. Geological Survey (USGS) Biological Resources Division, and USGS Water Resources Division. In 2001, the technical workgroup submitted a final summary report to the policy workgroup of the EPA Region 10 Water Temperature Guidance Project entitled *Technical Synthesis: Scientific Issues Relating to Temperature Criteria for Salmon, Trout, and Charr Native to the Pacific Northwest*.² Five technical summaries on the major physical and biological considerations for developing water temperature standards were developed to provide a scientific foundation for the project:

1. thermal effects on salmonid physiology (McCullough et al. 2001),
2. thermal effects on salmonid behavior (Sauter et al. 2001),
3. interactions between multiple stressors—thermal and other—affecting salmonids (Materna 2001),
4. thermal influences on salmonid distribution (Dunham et al. 2001), and
5. spatial and temporal variation in patterns of stream temperature (Poole et al. 2001a, b).

McCullough (1999) of the Columbia River Inter-Tribal Fish Commission, prepared a summary report for EPA Region 10, in which he reported that significant impacts to survival due to temperature regime can occur in all life stages. Sublethal impacts to life processes such as growth, survival, reproductive success, migration success, disease, feeding, territoriality, aggressiveness, swimming, and bioenergetics can cumulatively result in diminished survival and production of the population.

In December 2000, the WDOE Water Quality Program released a draft discussion paper and literature summary addressing temperature criteria that included recommendations for chinook, coho, and chum salmon and steelhead at critical life stages (WDOE 1999). The recommendations were based on a thorough review of the literature and laboratory data adjusted for application to natural waters.

The following sections discuss the scientific findings on the thermal effects and requirements of salmonids, in general and by individual species, and attempt to draw together materials from the documents listed above.

General Salmonid Data

Smoltification

High temperatures during the smolt phase can result in outright lethality, premature smolting, blockage of seaward migration, desmoltification, shifts in emigration timing resulting in decreased survival in the marine environment, and other stresses detrimental to fitness.

² Available in final form at

<http://yosemite.epa.gov/R10/WATER.NSF/6cb1a1df2c49e4968825688200712cb7/bd029c92a81bf25f88256a020072a8c7?OpenDocument>.

Temperatures that have been reported for impairing smoltification are above a range from approximately 12° to 15°C or more (McCullough et al. 2001).

Adult Migration

Thermal blockages to adult salmon migration have also been identified. Migration blockages occur consistently in the temperature range of 19° to 23°C (McCullough et al. 2001). For chinook and sockeye salmon and steelhead in the Columbia River 21.7° to 23.9°C has been cited as the temperature range blocking migration (Fish and Hanavan 1948).

Spawning

Elevated temperatures can cause migration delays in salmonids that alter timing of key processes, such as spawning, or can lead to stress, disease, bioenergetic depletion, or death. If salmonids such as fall chinook or coho are exposed to high temperatures just before or during spawning, gametes held internally in adults can be severely affected, resulting in a loss of viability that appears as poor fertilization rate or embryo survival. McCullough et al. (2001) concluded that egg mortality, alevin development, and egg maturation are negatively affected by exposure to temperatures above approximately 12.5° to 14°C. A spawning temperature range of 5.6° to 12.8°C (maximum) appears to be a reasonable recommendation for Pacific salmon, unless colder thermal regimes are natural in any tributary (McCullough et al. 2001).

Lethality

Analysis of lethal temperature suggested that a threshold of 26°C for annual maximum temperature is a signal of imminent risk of direct mortality (Sullivan et al. 2000). A site-specific analysis of duration of exposure when annual maximum temperature is between 24°C and 26°C is also recommended to ensure that duration/magnitude thresholds are not exceeded. These annual maximum temperature values are intended to apply to all salmon and trout species in natural rivers and streams in the Pacific Northwest (Sullivan et al. 2000).

Distribution

After a review of field studies for chinook salmon, steelhead, and rainbow trout, McCullough (1999) showed that the distributional limit of these salmonids corresponds approximately to a mean daily water temperature of 20°C and a maximum daily water temperature of 22° to 24°C. Hokanson et al. (1997) showed that water temperatures greater than 23°C, even for short periods (hours), result in Pacific salmon and trout moving into cold-water refugia. Eaton et al. (1995) proposed a higher (95th percentile) weekly mean temperature tolerance for chinook (24°C) and coho (23.4°C) salmon, when the fish temperature database matching system (FTDMS) approach was used to evaluate various distribution records for 30 common North American fish species. However, when using the FTDMS approach, the existence of thermal refugia provides a potential source of error (Eaton 1995), and leads to recommendations inconsistent with other studies. Many important salmon diseases become

virulent above 15.6°C, which makes the impact to population production potentially more severe, because as temperatures rise toward the limits to salmonid growth, the mortality rate increases. The balance may shift to zero net growth, even at lower temperatures. In general, juvenile salmonids appear to have final temperature preferences in the range 11.7° to 14.7°C (Ferguson 1958, Countant 1977, Jobling 1981, McCullough, 1999).

From a behavioral perspective, Sauter et al. (2001) suggest that water temperature affects some aspects of juvenile life-history patterns, including duration of freshwater rearing and outmigration timing. Elevated water temperatures inhibit gill ATPase activity, an enzyme that prepares juvenile salmonids for osmoregulation in seawater during emigration. Decreased gill ATPase activity is associated with loss of migratory behavior in anadromous juvenile salmonids. Spring water temperatures must not exceed 12°C for successful smoltification in steelhead. For spring chinook and coho this value is 15°C, and it may be higher for summer migrating fall chinook subyearlings (Zaugg and Wagner 1973). In addition, higher water temperatures and longer exposure to warm water increase the feeding rate of predatory species consuming juvenile salmonids. Interspecific competition also appears to play a role in the distribution and thermal preferences of juvenile salmonids (Sauter et al. 2001).

Swimming Speed

If water temperature is too high, changes in swimming speed can impair adult migration. Fish may refuse to migrate, migrate back downstream, or seek shelter in tributaries or other available cold-water refugia. Swimming speed is also vital to smolts' ability to maintain position in the current, control rate of descent, and avoid obstacles. In addition, high temperature can impair their ability to swim in quick bursts, which is necessary to avoid predators (McCullough et al. 2001).

Chemical Constituents

Many chemical constituents are affected by temperature. Most notably, dissolved oxygen (DO) decreases with increasing temperature. When fish experience temperature stress, they may also experience stress from low DO levels. McCullough (1999) showed that adult migration of chinook salmon can be impeded when temperature and DO requirements are not met. It is also well documented that the concentration of ammonia increases with increasing temperature (EPA 1985). Although there is no single pattern that explains the effects of temperature on the toxicity of pollutants to aquatic organisms, some evidence shows that temperature may change the rate of toxification under chronic exposures (Mayer et al. 1994). Since rising temperatures result in an increase in metabolic processes, gill ventilation must also rise proportionately (Heath and Hughes 1973). Black et al. (1991) showed that an increase in water flow over the gills, which may result from increased gill ventilation at increased temperature, resulted in rapid uptake of toxicants via the gills. Sublethal exposure to toxicants can reduce the upper lethal temperatures of fish, constricting the thermal tolerance zone (Paladino et al. 1980), and fish that are weakened by other causes may be much more sensitive to toxic chemicals (Jobling 1994). Temperature has been found to significantly increase the toxicity of some organic chemicals such as terbufors, trichlorfon, and 2,4 dinitrophenol (Howe et al. 1994), as well as some metals such as mercury

(MacLeod and Pessah 1973, Materna 2001). There is evidence that higher temperatures can help fish to withstand ammonia (cited in Sullivan et al. 2000).

Disease

Most fish diseases are exacerbated by higher water temperatures (Ordal and Pacha 1963) and can infect salmon at many life stages. Diseases associated with warm water in the Pacific Northwest are well documented. They include the bacterial infections *columnaris*, caused by *Flexibacter columnaris*; bacterial kidney disease (BKD), caused by *Renibacterium salmoninarum*; the bacterial pathogens *Aeromonas salmonicida*, *A. punctata*, *A. hydrophila*; and the protozoan parasite *Ceratomyxa shasta*. Evidence from Idler and Clemens (1959), Williams et al. (1977), Bouck et al. (1970) (as cited by EPA and NMFS 1971), and Ordal and Pacha (1963) indicates that temperatures of 16.7° to 20°C or higher, lead to infection of adult salmon with *columnaris*, even with exposure to low-virulence strains, and infection can occur at even lower temperatures with high-virulence strains. Evidence from Colgrove and Wood (1966) indicates that temperatures between 13.9° and 15.6°C constitute a transitional temperature region below which recovery from *columnaris* after infection could occur, and above which infection and mortality increase. Laboratory and field studies by numerous investigators show that infection and mortality by *columnaris* disease were negligible at temperatures $\leq 12.8^\circ\text{C}$, but temperatures $\geq 15^\circ\text{C}$ produced significantly increased mortalities. Not only do juvenile survival rates decrease with increasing temperature, but Fryer and Pilcher (1974) also showed that time to death decreases with increasing temperature for juvenile chinook salmon, coho salmon, and steelhead (Materna 2001).

BKD is also a prevalent disease in which temperature has been shown to have an effect on the mortality of infected salmonids. In an experiment involving infected sockeye, coho, and steelhead over a range of temperatures from 4° to 20.5°C, the highest mortalities due to BKD occurred at 12.2°C, with declining mortalities at higher or lower temperatures (Fryer and Sanders 1981, as cited in Chapman et al. 1991). It is important to remember that elevated temperatures do not increase mortality from all salmonid diseases; in fact, some diseases have higher mortalities at temperatures well within an otherwise optimal range.

Groberg et al. (1978) studied the relationship of water temperature to infections of coho salmon, chinook salmon, and steelhead with *A. salmonicida* and *A. hydrophila*. Among the three salmonid species, at 3.9° and 6.7°C, mortality in fish infected with *A. salmonicida* varied from 2 to 26%; at 20.5°C, 93 to 100% died within 2 or 3 days; at 6.7°C or lower survival was 12 to 23 days. Results from experiments with *A. hydrophila* gave similar results. At 20.5°C, mortality ranged from 64 to 100%; at 9.4°C or below, no deaths occurred.

General Recommendations for Salmonids

The Pacific Northwest Salmon Habitat Indicators Work Group (PNWSHIWG). (1998) identified maximum water temperature as an influence on salmonid migration patterns, development of eggs to alevins, fry emergence, metabolism, behavior, susceptibility to parasites and disease, and mortality. Water exceeding 20°C was categorized as causing “severe impairment.”

Chinook Salmon Data

Incubation and Early Fry Development

Based on the works of Donaldson (1955), Garling and Masterson (1985), Seymour (1956), Eddy (1972, as cited in Raleigh et al. 1986), Burrows (1963), Baily and Evans (1971), Heming (1982), Heming et al. (1982), the following temperatures are strongly suggested to provide optimum conditions for incubation and early fry development for chinook salmon. Constant temperatures above 9° to 10°C may reduce the survival of embryos and alevins. Temperatures of 11° to 12°C can still result in good survival, however the results are consistently less than what is produced at lower temperatures (McCullough et al. 2001). Incubation temperatures from 13.9° to 19.4°C have been associated with complete mortality while significant mortality (over 50%) has been noted at constant incubation temperatures from 9.9° to 16.7°C (Hicks 2000).

Juvenile Rearing and Growth

Optimal rearing temperatures at natural feeding regimes are in the range of 12.2° to 14.8°C for chinook salmon (Hicks 2000). Banks et al. (1971, as cited by Garling and Masterson 1985), Clarke and Shelbourn (1985), Brett et al. (1982), and Marine (1997) reported optimum growth temperatures determined from feeding on full rations that range from 14.8° to 20°C. Ration size in the laboratory and food supply in nature can have significant effects on optimal temperatures for rearing, and this complication is one that must be kept in mind when evaluating temperature effects in eventual monitoring and evaluation efforts. Feeding rates below the satiation level typical of field situations are associated with reduced optimum growth temperatures (Elliott 1981). Brett et al. (1982) reported an optimal growth temperature of 19°C for chinook maintained in the laboratory at maximal daily ration, but that growth rates in the field corresponded to a projected feeding level of 60% of maximal daily ration, which translated to an optimal growth temperature of 14.8°C for the field population.

Smoltification

Although data on temperature impairment of smoltification is incomplete, the existing literature suggests that temperatures should be generally maintained below 12° to 13.8°C during outmigration of chinook salmon smolts (Hicks 2000). The temperature threshold for impairment of smoltification was found to be 12°C by Zaugg (1981) in spring chinook yearlings, while Marine (1997) found it to occur at 17° to 20°C in fall chinook subyearlings.

Adult migration

Immigrating spring chinook salmon in the Willamette River have experienced thermal blockages at 21° to 22°C (at oxygen 3.5 mg/l) (Alabaster 1988). A temperature of 21°C blocked migration of spring chinook salmon in Clearwater, Idaho, (Stabler 1981) as well as summer chinook salmon (Stuehrenberg et al. 1978, as cited by Dauble and Mueller 1993) of the Snake

River. A temperature of 21.1°C blocked spring chinook in the Tucannon River (Bumgarner et al. 1997), and fall chinook in the Sacramento River were blocked at 19° to 21°C (oxygen ~ 5mg/l) (Hallock et al. 1970).

Spawning

The following authors reported spawning temperature ranges in daily average temperatures (DAT) for chinook salmon. For spring chinook salmon, Olson and Foster (1955) reported 4.4° to 17.8°C. For fall/summer chinook, Raleigh et al. (1996, cited in ODEQ 1995) reported 5° to 13.4°C. The majority of the temperature observations reviewed in Hicks (2000) cited a maximum spawning temperature below 14.5°C for chinook salmon.

Lethality

For chinook salmon, the upper incipient lethal temperature (UILT) has been recorded at 25.1°C (acclimation temperature 20° and 24°C) by Brett (1952), and 24.9°C (acclimation temperature 21.1°C) by Orsi (1971).

Behavior

For subyearling spring chinook salmon in the Dungeness River, Brett (1952) found the acute preference temperature to be 12° to 13°C at all acclimation temperatures and the mean final preference temperature was 11.7°C. Sauter (1996) found that spring chinook salmon smolts on unlimited ration have a final temperature preference of 16.7°C and Spigarelli (1975) reported that adults prefer a field temperature of 17.3°C. For fall chinook salmon, Sauter (1996) found parr to prefer a mean 16.7°C, while advanced smolts preferred 10.9°C.

Recommendations for Chinook Salmon

Incubation and Early Fry Development

McCullough et al. (2001) recommended that temperatures be maintained below 12°C for incubation and fry development, and Hicks (2000) recommended an adjusted 7-day average of the daily maximum temperatures (7-DAM) of 11° to 12°C at the time of fertilization of chinook salmon eggs. Both McCullough et al. (2001) and Hicks (2000) recommended and that individual daily maximum temperatures (1-DM) of 13.5° to 14.5°C are required to provide optimal protection from fertilization through early fry development.

Growth

McCullough (1999) suggested using the growth optimum of 15.6°C for spring chinook salmon as the temperature standard, because temperatures lower than this cause no reduction in survival while temperatures higher than this begin to reduce growth and lead to increasing

mortality rates. A synthesis of evidence from Bisson and Davis (1976) (as cited by Garling and Masterson 1985), Brett et al. (1982), Marine and Cech (1998), Wilson et al. (1987), Reiser and Bjornn (1979), and Brett (1952), lead McCullough et al. (2001) to recommend an optimum production temperature zone of 10.0° to 15.6°C. Adjusting laboratory temperatures to naturally fluctuating stream temperatures, Hicks (2000) recommended that a 7-DAM of 14.2° to 16.8°C during the peak of summer provides for optimal growth conditions for chinook salmon.

Adult Migration

Hicks (2000) recommended that daily maximum temperatures should not exceed 20° to 21°C in order to prevent migration blockage of adult chinook salmon.

Lethality

Hicks (2000) recommended that to protect fish from acute lethality, daily maximum temperatures not exceed 22°C. In addition, he recommended that thermal plumes should not be allowed such that fish could become even briefly exposed to water warmer than 30° to 32°C.

Coho Salmon Data

Incubation and Early Fry Development

From the studies of Dong (1981), Tang et al. (1987), Murray -and McPhail (1988), Velsen (1987), and Davidson and Hutchinson (1938) (as cited in Sandercock 1991), it is relatively clear that egg survival for coho salmon is consistently best at constant temperatures of 2.5° to 6.5°C, but may still be acceptable for many stocks at temperatures of 1.3° to 10.9°C. Alevin and fry survival and health may be best at constant temperatures of 4° to 8°C, but survival may remain acceptable up to 10.9°C. A constant 12°C may form the upper threshold for optimal development of coho salmon eggs and alevin (McCullough et al. 2001, Hicks 2000).

Flett et al. (1996) investigated the cause of low survival to hatch of embryos (42%) of coho salmon from the Fairview, Pennsylvania, stock in Lake Erie in 1988. It was proposed that the low survival was due to delayed oocyte maturation, ovulation, and vent maturation. Flett et al. (1996) suggested this was caused by exposure of the salmon to warm water (above 20°C in his Fairview stock and 2° to 4°C higher than in the Simcoe stock, which showed no such impairments) during late ovarian maturation and migration.

Juvenile Rearing and Growth

Most literature shows that juvenile coho salmon are not particularly sensitive to stream temperatures and generally suggests maximum temperatures between 9.4° and 14.4°C as optimal. However, Everson (1973, as cited by Sullivan et al. 2000) found that, depending on food availability, growth optima occur at 15°C. Average or constant temperatures of 12° to 15°C probably best characterize optimal rearing conditions (Hicks 2000). The Sustainable Ecosystems Institute review (Sullivan et al. 2000) suggested 12° to 17°C as an acceptable temperature range.

Smoltification

Both Zaugg and McLain (1976) and Adams et al. (1975) reported the temperature threshold for impairment of smoltification of coho salmon to be 15°C.

Spawning

It has been reported that spawning activity in coho salmon may typically occur in the range of 4.4° to 13.3°C (Hicks 2000), although Bell (1973) suggested that temperatures should be within the range of 7.2° and 15.6°C for successful spawning of coho salmon. Bell (1991) reported a DAT of 10° to 12.8°C for spawning coho salmon.

Lethality

For coho fry, Brett (1952) reported UILT (the temperature at which 50% of the population is dead after indefinite exposure) at 25.0°C (acclimation temperatures of 20° and 23°C). Konecki et al. (1995) tested juvenile coho salmon fry critical thermal maximum (CTM, the species-specific temperature at which a fish loses equilibrium and dies, which depends on acclimation temperature). Mean CTMs from three populations captured in the field in Washington State were 28.2°, 29.1°, and 29.2°C, which exceed published data from some laboratory tests for juvenile coho (Beschta et al. 1987, DeHart 1975, McGeer et al. 1991). The population from a relatively cool stream had a lower CTM than two populations from warmer streams. After three months in the laboratory under constant temperature regimes the CTMs no longer differed. This indicated that the population-specific differences resulted from different acclimation regimes rather than from genetic adaptation. Constant exposure to temperatures of 22° to 23°C poses a risk of causing direct lethality to juvenile coho salmon (Hicks 2000).

Behavior

For subyearling coho salmon, Brett (1952) reported a range of 12° to 14°C for their temperature preference, which is affected by acclimation temperature. Konecki et al. (1995) reported 11.6°C (range 7° to 21°C) and 9.9°C (range 6° to 16°C) for the final temperature preference (species-specific value that may be influenced by feeding level) for subyearling coho salmon in two different creeks. Reutter and Herdendorf (1974) reported that adult coho have a final preference temperature of 11.4°C, while Spigarelli (1975) reported a preferred field temperature of 17.3°C.

Swimming Speed

Brett et al. (1958) investigated the effect of temperature on the cruising speed of young coho salmon. Cruising speeds of subyearling and yearling coho were determined for acclimation temperatures ranging from 1° to 24°C. Optimum cruising speed for juvenile coho occurred at 15°C.

Disease

Groberg et al. (1983) studied the effects of water temperature on infection by the predominantly marine pathogen *Vibrio anguillarum* in juvenile coho salmon at seven water temperatures range from 3° to 21°C. More rapid death and higher mortality were observed at the elevated water temperatures. Growth rates of *V. anguillarum* were directly related to temperature.

Recommendations for Coho Salmon

Incubation and Early Fry Development

Adjusting laboratory temperatures to naturally fluctuating stream environments resulted in a recommendation of a 7-DAM of 9° to 12°C to fully support the pre-emergent states of coho salmon (McCullough et al. 2001, Hicks 2000).

Juvenile Rearing and Growth

Adjusting for a naturally fluctuating stream environment resulted in a recommendation of 14° to 17°C for the 7-DAM to fully protect juvenile coho salmon rearing (Hicks 2000).

Sullivan et al. (2000) developed and used a bioenergetics-based approach to evaluate salmon growth in relation to environmental temperature, and to suggest sublethal temperature thresholds for coho salmon. An upper threshold for the 7-DAM temperature of 16.5°C was found to be appropriate, assuming a 10% reduction in growth represents an appropriate risk level (Sullivan et al. 2000).

Lethality

Subtracting a 2°C safety factor resulted in a recommendation of 20° to 21°C to avoid direct lethality to coho salmon (Hicks 2000).

Chum Salmon Data

Incubation and Early Fry Development

Based on the works of Murray and Beacham (1986), Beacham and Murray (1985), and Zinichev and Zotin (1988), constant incubation temperatures from 4° to 12°C commonly produce excellent incubation results for chum salmon; however, some researchers have noted that less-than-optimal survival occurs at the edges of this range. Both McCullough et al. (2001) and Hicks (2000) suggested that constant initial incubation temperatures of 8° to 10°C would be most consistently optimal for chum salmon.

Juvenile Rearing

Optimal rearing occurs between about 13° to 14.5°C (Hicks 2000).

Spawning

The Independent Scientific Group (1996) reported an average range of 8° to 13°C for spawning, while Hicks (2000) reported that chum salmon most consistently spawn within a range of 7° to 10.5°C.

Lethality

Brett (1952) reported the UILT for chum salmon fry at 23.7° and 23.8°C (acclimation temperature 20° and 23°C, respectively). Hicks (2000) stated that significant lethality to chum salmon can result from constant exposure to 22° to 23°C.

Behavior

Brett (1952) reported that juvenile subyearling chum salmon have an acute preference temperature of 12° to 14°C at all acclimation temperatures and final preference temperature of 14.1°C. Groot and Margolis (1991) reported adult migrant chum have an acute preference temperature of 7° to 11°C.

Recommendations for Chum Salmon

Incubation and Early Fry Emergence

Hicks (2000) recommended that the 7-DAM should not exceed 10° to 12°C for fertilization through fry emergence.

Lethality

With the 2°C safety factor, it was recommended that daily maximum temperatures should not exceed 20° to 21°C to prevent direct lethality to chum salmon. In addition, fish should not be exposed even briefly to temperatures greater than 33° to 34°C (Hicks 2000).

Steelhead Data

Incubation and early fry development

Considering the works of Fuss (1998), Bell (1986), Rombough (1988), and Redding and Schreck (1979), it appears that an optimal constant incubation temperature occurs below 11° to 12°C for steelhead (McCullough et al. 2001).

Juvenile Growth

Optimal growth for juvenile steelhead occurs in the range of 14° to 15°C (Hicks 2000); although in a laboratory setting, Wurtsbaugh and Davis (1977) found that steelhead growth could be enhanced by temperature increases up to 16.5°C.

Smoltification

For steelhead, Hoar (1988) reported temperatures higher than 13°C, Adams et al. (1975) reported higher than 12.7°C, Zaugg et al. (1972, as cited by Zaugg and Wagner 1973) reported higher than 13.6°C and Zaugg (1981) reported 12°C as upper thresholds for impairment of smoltification.

Adult migration

Strickland (1967, as cited by Stabler 1981) reported 21°C as the temperature blocking adult steelhead migration in the Snake River.

Spawning

For steelhead, Bell (1991) reported a daily average temperature range of 10° to 12.8°C for spawning.

Behavior

For subyearling steelhead in the South Umpqua River, with food available, the preferred temperature was 15.0°C and 17.8°C for yearlings (Roper and Scarnecchia, 1994).

Nielsen et al. (1994) studied steelhead use of thermally stratified pools in Northern California streams. It was observed that 65% of the juvenile steelhead in Rancheria Creek moved into thermal refugia—in the form of adjacent stratified pools—during periods of high ambient stream temperatures of 23° to 28°C. Just before moving into these pools, fish showed a decline in foraging behavior and increased agonistic activity. On the Middle Fork Eel River, summer-run steelhead adults were found in deep stratified pools throughout the summer, when midday ambient stream temperatures ranged from 26° to 29°C; these cold water pockets were on average

3.5°C cooler than the stream. Where stream temperatures reached upper incipient lethal levels, these thermally stratified pools provided refuge habitat for significant numbers of young-of-the-year, yearling, and adult steelhead.

Recommendations for Steelhead

Incubation and Early Fry Development

Based on the literature and adjusting for a naturally fluctuating river environment, the recommendation of 13.5° to 14.5°C for the single daily maximum temperature from fertilization through hatching was made by Hicks (2000).

Juvenile Rearing and Growth

The adjusted value for recommendation to fully protect juvenile rearing of steelhead was 16° to 17°C (Hicks 2000). Sullivan et al. (2000) recommended the upper threshold for the 7-DAM temperature of 20.5°C for steelhead, assuming that a 10% reduction in growth is an acceptable risk level.

Smoltification

Hicks (2000) adjusted constant temperature ranges to the fluctuating stream environment, and recommended a 7-DAM of 13.3° to 14.3°C for emigrating steelhead smolts.

Adult Migration

Based on the consistency of several studies, Hicks (2000) recommended that temperatures remain lower than 21° to 22°C (1-DAM) to prevent thermal barriers to migrating steelhead, and that water in which steelhead migrate or hold not exceed a 7-DAM of 16° to 17°C.

Lethality

Hicks (2000) recommended that daily maximum temperatures remain below 19° to 20°C to prevent directly lethal conditions to steelhead.

Draft Criteria

The EPA (2001) recommends that temperature-limit criteria be based upon upper optimal physiological temperature preferences known to support requisite biological processes of recognized salmonid life-history stages. Moreover, the EPA (2001) recommends that the criteria be based on guilds of salmonids, taking the spatio-temporal use of the landscape by guild members into account. For the Lower Columbia and Willamette Rivers, the guild present is the cold-water guild. Along with the population growth and abundance criteria being developed by

the WLC-TRT and reported elsewhere, we suggest that temperature-based criteria for the delisting of threatened and endangered salmonids under the Endangered Species Act, written to be consistent with EPA upper optimal temperature values, consist of two parts:

1. A requirement that in order to delist salmonids in the Willamette and Lower Columbia River domain, the 7-DAM temperature maxima within the habitat of a given evolutionarily significant unit (ESU) must not be increasing over the course of 20 years. This requirement of thermal nondeterioration is intended to complement requirements for nondeterioration in population growth rates and abundance. Prior to delisting, data will need to be collected to show with high confidence that the slope of the observed temperature trend is less than or equal to zero. The 20-year period provides a long enough data set to avoid being confounded by temperature oscillations driven by the Pacific Decadal Oscillation (PDO) (Anderson 1998, Chao et al. 2000), which has considerable effects on climate in the Pacific Northwest.
2. Upper optimal temperature criteria be adopted, above which delisting cannot occur regardless of whether or not the direction of change is nondeteriorating. We suggest the following as temperature maxima above which delisting cannot occur for chinook, coho, chum salmon, and steelhead:

	7-Day-Average Maximum Daily Temperatures	Weekly Mean Temperatures
Spawning and incubation	13°C (55°F)	10°C (50°F)
Juvenile rearing	16°C (61°F)	15°C (59°F)
Adult migration	18°C (64°F)	16°C (61°F)
Smoltification except steelhead	16°C (61°F)	15°C (59°F)
Steelhead smoltification at fourth-level HUC ^a watershed	14°C (57°F)	12°C (54°F)

^a HUC = hydrologic unit code

For all these criteria, the significant challenge of defining the spatiotemporal range over which they should be applied remains. Those spaces occupied by threatened and endangered salmonids need to be regulated at the times of year that sensitive life stages are present, and defining the bodies of water involved and the times to apply the standards requires additional consideration and research. The concept of assessing thermal potential being developed by the EPA involves modeling the characteristics of bodies of water in order to determine whether a distribution of temperatures sufficient to support salmonids (and other beneficial uses) can be attained (see “Public Review Draft” at <http://www.tboys.com/chalk2.htm>). Using this use-attainability analysis, as prescribed by the existing Clean Water Act, there is no obligation to provide unattainable conditions (in this case to apply the temperature criteria). It may be that as the EPA develops concepts related to use-attainability analysis (including appropriate model selection, sensitivity analysis, and determination of an acceptable level of anthropogenic degradation), it will begin to converge with the efforts of the WLC-TRT and others involved in salmon recovery planning to define spatio-temporal ranges over which maximum temperature criteria will be applied. Some amelioration of the difficulties posed by this challenge may be provided by the multiple runs and species that broaden the times of concern beyond brief periods

for many subbasins. Once the spatio-temporal pattern to apply these standards has been defined for any ESU, exemptions to the above temperature maxima for specific bodies of water can still be proposed based on historical acclimation to higher temperatures. Such an exemption would require physiological or population-level evidence that higher temperature maxima would not harm the fish native to that area, or that cold-water refugia are plentiful and provide the circumstances required for salmonid survival. Evidence required for an exemption would need to include the density, size, and duration of thermal refugia; data indicating that the distribution of refugia in space and time is adequate to be protective of the salmonids (Ebersol 2002); and field and laboratory studies providing strong evidence of physiological acclimation for the existing local population whose habitat is under consideration.

The guild-based temperature criteria are supported by the data collected for the salmonid species at the Willamette and Lower Columbia Rivers. The relationships between individual species data and the guild-based criteria are described briefly in the following paragraphs.

Spawning and Incubation

The 10°C weekly mean temperature criterion is consistent with the upper temperature range for optimum survival of chinook salmon embryos and alevins (Raleigh et al. 1986) and is within reported temperature ranges for successful spawning (Olson and Foster 1955, Raleigh et al. 1986), although the majority of spawning observations reported by Hicks (2000) recommended maximum temperature values for chinook salmon consistent with the proposed criterion of 13°C.

For coho salmon, the weekly mean temperature criterion of 10°C is at the upper end of their acceptable incubation temperature range (McCullough et al. 2001, Hicks 2000). This criterion is within the acceptable range of coho spawning temperatures (Hicks 2000, Bell 1973).

For chum salmon, the 10°C weekly mean temperature criterion is within their safe temperature range for spawning (Hicks 2000) and incubation (McCullough et al. 2001, Hicks 2000).

Steelhead spawning occurs at temperatures within the range protected by the 10°C (weekly mean temperature criterion), as does their early fry development (McCullough et al. 2001).

Juvenile Rearing

The 15°C weekly mean temperature criterion is at the upper edge of optimal rearing temperatures for chinook with a natural feeding regime (Hicks 2000).

The 15°C weekly mean temperature criterion is at the upper end of the temperature range providing optimal rearing conditions for coho salmon (Hicks 2000).

The 15°C weekly mean temperature criterion is slightly above the optimal range for chum rearing reported by Hicks (2000).

Optimal growth temperatures for juvenile steelhead are in the range of 14° to 15°C (Hicks 2000), although in a laboratory setting slightly higher temperatures were associated with a food supply in excess of that characteristically available in nature (Wurtsbaugh and Davis 1977).

Smoltification

The extreme variability of habitat use by steelhead makes establishing a temperature criterion for their smoltification challenging. The 12°C proposed for a weekly mean temperature at the fourth-level hydrologic unit (HUC) watershed is consistent with Zaugg and Wagner's (1973) gill ATPase activity data. Weekly mean temperature values of 15°C proposed as criteria for other salmonids are well above the values having excessive physiological consequences for steelhead (Zaugg and Wagner 1973). The results of Adams et al. (1975) and Hoar (1988) support this lower criterion for steelhead.

The weekly mean temperature criterion of 15°C may be more protective of fall chinook salmon (Marine 1997) than spring chinook (Zaugg 1981). Hicks (2000) found that temperatures above 13.8°C did produce smoltification impairment in chinook.

For coho salmon the 15°C weekly mean temperature criterion is at the threshold temperatures that cause smoltification impairment.

Adult Migration

The proposed maximum temperature criterion of 16°C is within the safe range proposed for chinook temperature maxima by Hicks (2000) and seems protective for coho and chum salmon survival during adult migration as well (Hicks 2000). Adult migrant chum have a somewhat lower temperature preference of 7° to 11°C (Groot and Margolis 1991). Adult steelhead migration is not blocked until 21°C (Strickland 1967, as cited by Stabler 1981). Steelhead have been reported to make use of deep stratified pools as thermal refugia when midday ambient stream levels ranged above their tolerance limits (Nielsen et al. 1994).

Framework for Temperature Criteria

Salmonid survival and recovery will require more than the attainment of these temperature goals. A rich data set shows that in terms of thermal tolerances, disease resistance and physiological adaptation in general salmonid stocks native to specific bodies of water may be better adapted to local conditions than are other members of their species. However, in many populations the genetic modification due to hatchery operations may significantly reduce the present levels of local adaptation. Definitive criteria for salmonid recovery should eventually define ways to incorporate spatio-temporal variability into them in a realistically complex fashion and have as their eventual goal a process that realigns the curves of current environmental variables so that they overlay historic conditions rather than simply acting as a floor or ceiling. The challenge of this task is exacerbated by the multiple salmonid life stages that will need to be identified in their distribution over space and time and monitored. It is crucial that along with the attainment of habitat goals, historical salmonid populations be identified and recovered in a way that maintains them in the milieu suitable for their survival. Hatchery operations may need to be adjusted to serve this goal. Salmonid harvest patterns and hydro operation management may both need to take the significance of both environmental recovery and the relationships of specific genetic stocks to their native rivers into account (this study

provides no information on hatcheries, hydro, or harvest). The complexity of any solution to the problem of salmonid survival will need to balance all of these considerations while achieving temperature regimes suitable for the persistence of the salmon ESUs.

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